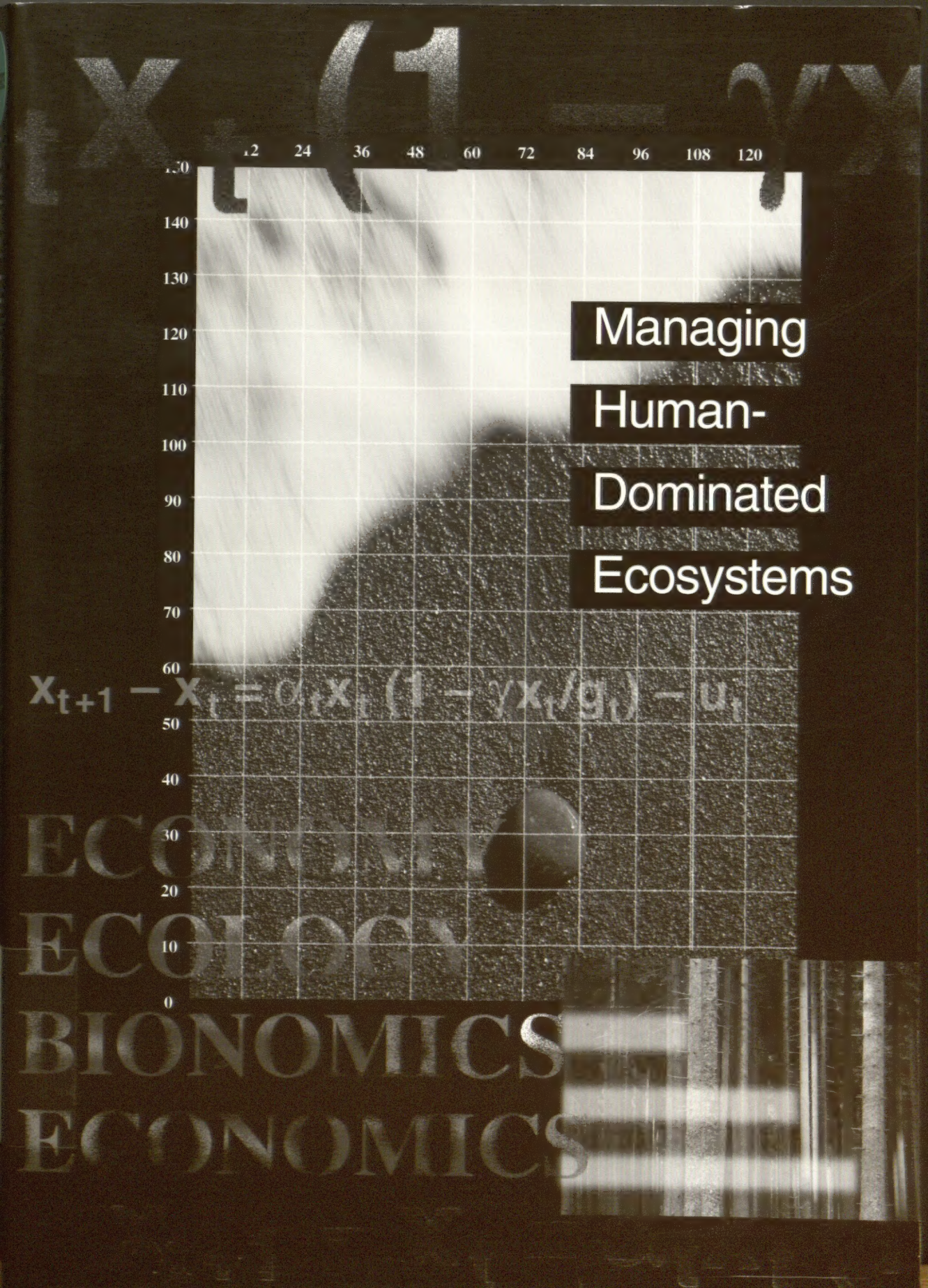
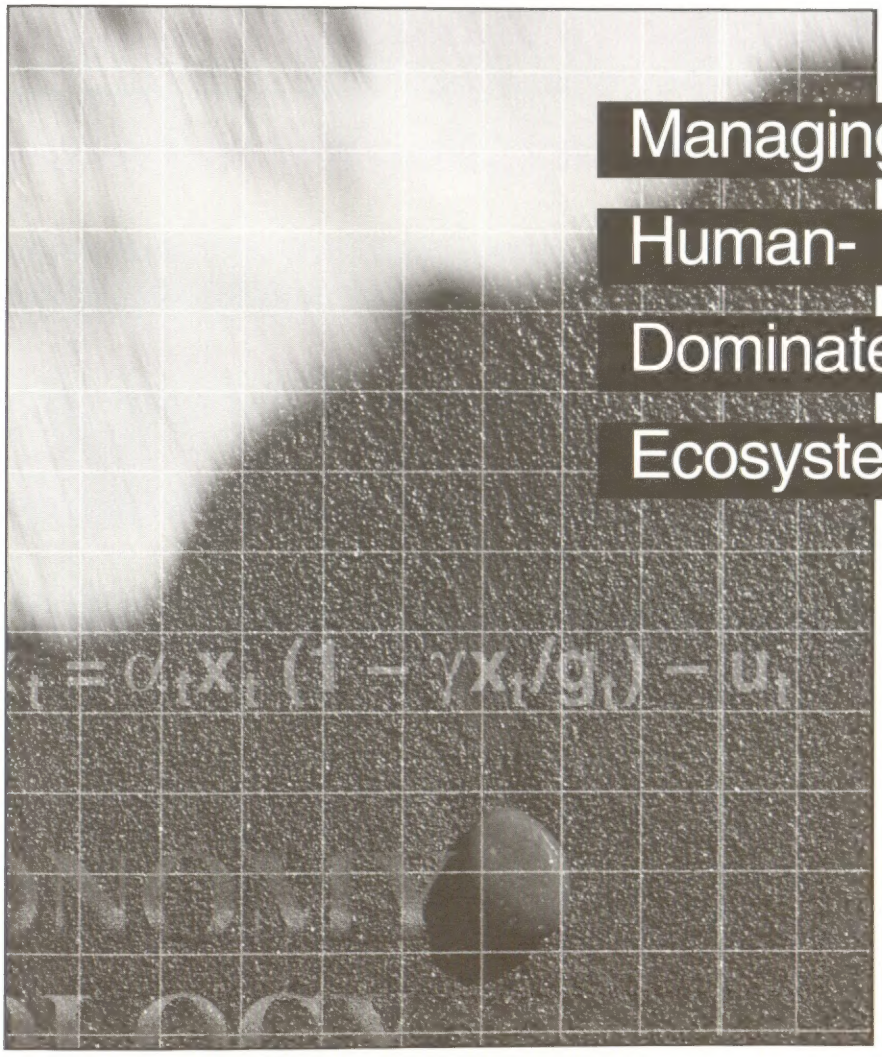
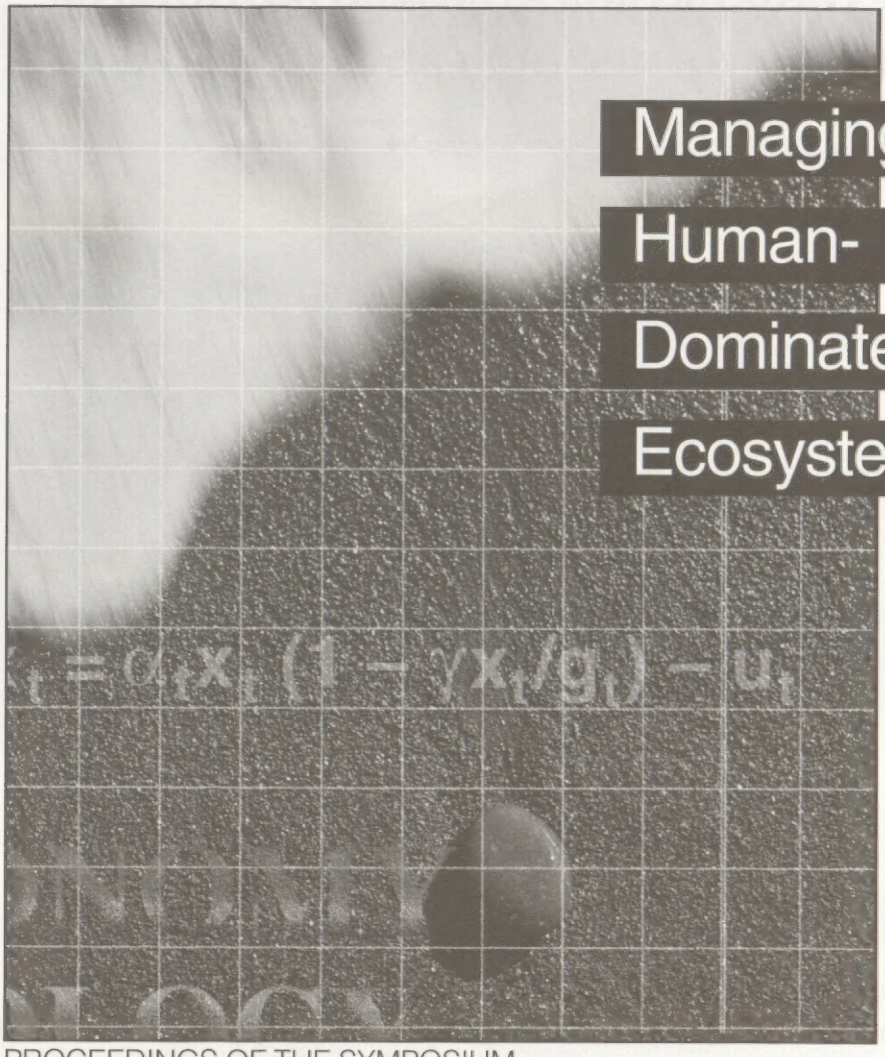

12 24 36 48 60 72 84 96 108 120
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Managing
Human-
Dominated
Ecosystems
$X_{t+1} - X_t = \alpha_t X_t (1 - \gamma X_t / g_t) - u_t$
ECONOMICS
ECOLOGY
BIONOMICS
ECONOMICS


Managing
Human-
Dominated
Ecosystems



Managing Human-Dominated Ecosystems

PROCEEDINGS OF THE SYMPOSIUM
AT THE MISSOURI BOTANICAL GARDEN,
ST. LOUIS, MISSOURI, 26–29 March 1998

Organizing Committee

Graciela Chichilnisky, The Columbia Earth Institute, Columbia University (Chair)
Gretchen C. Daily, The Center for Conservation Biology, Stanford University
Paul Ehrlich, The Center for Conservation Biology, Stanford University
Geoffrey Heal, The Columbia Earth Institute, Columbia University
James S. Miller, Missouri Botanical Garden

MONOGRAPHS IN SYSTEMATIC BOTANY FROM THE MISSOURI BOTANICAL GARDEN

Volume 84

ISBN 0-915279-85-1
ISSN 0161-1542

EDITOR
Victoria C. Hollowell

MANAGING EDITOR
Amy Scheuler McPherson

ASSOCIATE EDITOR
Diana Gunter

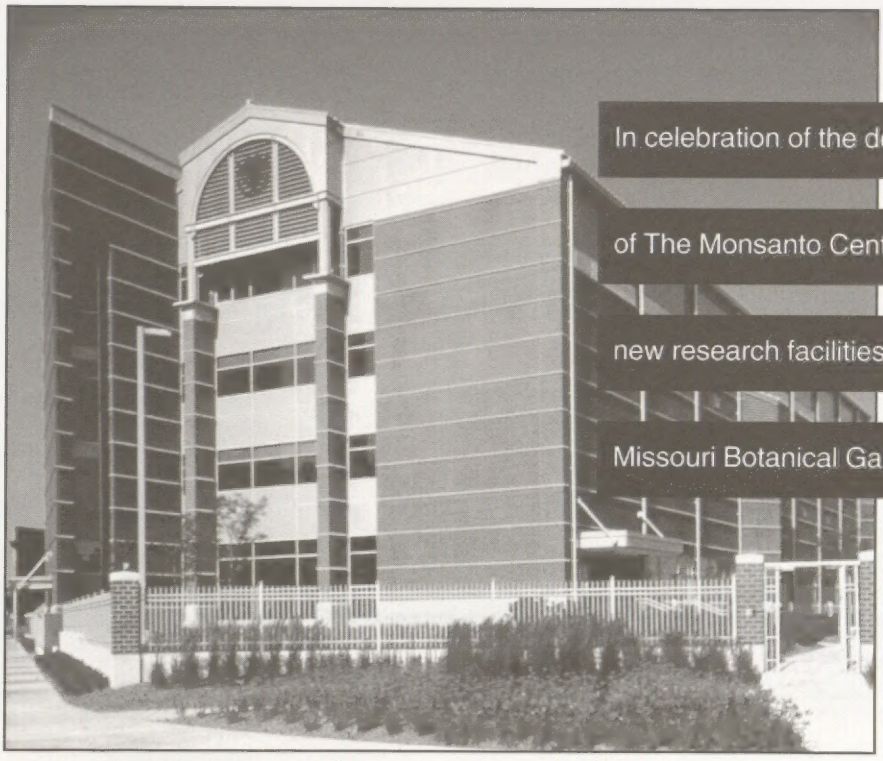
TEXT FORMATTER
Barbara Mack

SENIOR SECRETARY
Aida Kadunic

Typeset by Composition and Design Services, Brampton, Ontario, Canada.

Copyright © 2001 by Missouri Botanical Garden Press.
P.O. Box 299
St. Louis, Missouri 63166-0299, U.S.A.
All rights reserved.
Printed in U.S.A.

www.mobot.org



In celebration of the dedication

of The Monsanto Center,

new research facilities at the

Missouri Botanical Garden

Table of Contents

Preface *ix*

Introduction
by Victoria C. Hollowell and Peter H. Raven *x*

Ecosystem Services: Characterization of the Function and Principal Threats

How Changes in Biodiversity May Affect the Provision of Ecosystem Services
by Shahid Naeem 3

Sustainable Habitation in the Savannas
by Brian Walker 34

Forest Ecosystems Functioning, Threats and Value: Mexico as a Case Study
by Rodolfo Dirzo 47

A Note on the Interaction Between Environmental and Economic Factors
by Murray Weidenbaum 65

The Contribution of Economics to Ecosystem Preservation: Far Beyond Monetary Valuation
by Thomas Michael Power 69

Integrating Ecosystem Services into Environmental Law
by James Salzman 77

The Capture and Distribution of Ecosystem Service Values

Valuing Ecosystem Services
by David A. Starrett 85

Valuing Forests
by Bengt Kriström 97

The Value of Water and Watersheds: Reconciling Ecological and Economic Externalities
by Edward B Barbier 117

The Monetary Valuation of Biodiversity: Promises, Pitfalls, and Rays of Hope
by John M. Gowdy 141

PATHS TO SUSTAINABLE DEVELOPMENT: THE KNOWLEDGE REVOLUTION

The Global Environment in the Knowledge Revolution
by Graciela Chichilnisky 153

SYNTHESIS AND FUTURE DIRECTIONS

Capturing the Value of Ecosystem Services to Protect Biodiversity
by Walter V. Reid 197

Ecological Economics in Poor Countries: The Current State and a Program for Improvement
by Partha Dasgupta 226

Industrial Ecology
by Iddo K. Wernick 235

Markets and Sustainability
by Geoffrey Heal 249

A VIEW FROM THE SOUTH

The Environmental Impact of Globalization on Latin America: A Prospective Approach
by Gilberto C. Gallopín and Graciela Chichilnisky 271

Property Rights of Public Natural Resources: A New Legal Alternative for Developing Countries
by Fernando Gabriel Irrera, Mariano Javier Oneto, and Jorge Rabinovich 304

Economics and Environment in Developing Countries: Non-Traditional Environmental Policy Instruments
by Carlos Galperín and Jorge Rabinovich 312

Environment and Economics from the Perspective of the South
by Jorge Rabinovich 328

Preface

In the fresh St. Louis spring of 1998, a group of scientists remarkable for their breadth of talent across both economic and ecological lines convened at the Missouri Botanical Garden for a symposium entitled "*Managing Human-Dominated Ecosystems.*" This assemblage of social and natural scientists hallmarked the dedication of the Garden's new Research facilities located in The Monsanto Center. Co-sponsored by the Missouri Botanical Garden, Stanford University's Center for Conservation Biology, and The Columbia Earth Institute at Columbia University, this symposium received funding from the Albert P. and Blanche Y. Greensfelder Foundation and the Monsanto Fund. Handling the responsibilities of the organizing committee were Gretchen Daily and Paul Erhlich from Stanford University, Graciela Chichilnisky and Geoffrey Heal from Columbia University, and Jim Miller from the host institution in St. Louis. This scholarly exchange extended over four days of intense meetings and productive exchanges lasting from Thursday, March 26 through Sunday, March 29. The symposium centered on four major topics, with the first being *Ecosystem Services and their Function* with the subheading "an educational session for social scientists to consider principal threats." Manuscripts on ecosystem services include the first six in this volume from Naeem, Walker, Dirzo, Weidenbaum, Power, and Salzman. Moving on to a session targeting natural scientists, the second topic on Friday covered *The Capture and Distribution of Ecosystem Service Values.* Four authors (Starrett, Kriström, Barbier, and Gowdy) explore this natural valuation in a second group of articles. As the third focus on Saturday, *The Path to Sustainable Development: The Knowledge Revolution* was ably examined by individual speakers and by round-table discussions including Paul and Anne Ehrlich, Karl-Goran Maler of the Beijer Institute, and others. Graciela Chichilnisky presented her synthesis toward sustainability. Finally on Sunday, Peter Raven brought synthesis, pointing to *Future Directions* as the fourth topic. Included in this book from this evaluation across ecological and economic arenas are papers by Reid, Dasgupta, Wernick, and Heal. In a creative corollary, a fifth topical group of manuscripts joins the proceedings of this symposium. These offer a *View from the South* for developing countries and are contributed again by Chichilnisky and joined by authors Gallopín, Irrera, Oneto, Rabinovich, and Galperín.

Introduction

Victoria C. Hollowell and Peter H. Raven
Missouri Botanical Garden
P.O. Box 299
St. Louis, Missouri 63166-0299

We know unequivocally that biodiversity is globally diminishing through anthropogenic agency. What is more difficult to grasp is the impact of declining biodiversity on Earth's ecosystems and biogeochemical processes. *Shahid Neem* models this, simulating relationships between diversity and ecosystem functioning. One example is provided by *Brian Walker*, who explores savannas, their goods and services, and intertwines their floristic, faunal, and human factors. The relationship between savannas and subsistence agriculture is dynamic. These seasonal grasslands are complex adaptive systems, and it is in this light that our environmental and economic behaviors should interpret them to best advantage. In another example, *Rodolfo Dirzo* surveys the conflicting tension between biodiversity conservation and commercial exploitation with current land use patterns from Mexican forests. He advocates the issue of compensation to local communities as crucial to sustainable forest management for countries with similar profile.

Murray Weidenbaum cautions that a country's economic health drives its environmental awareness. A stark relationship exists between per capita income and manifest levels of pollution, in demonstration of the urgency for a greater common ground between advocates of environmental improvements and those concerned with economic efficiencies. Any potential contribution of economists to ecosystem integrity must move beyond mundane estimates of monetary value, as noted by *Thomas Michael Powers*. Equally parcel to the equation are human systems of incentive, and alternative pursuits of goals, as these evolve into public policy. Decentralization toward an environmentally responsible decision-making, as well as a cost-efficient, one must consider the full spectrum of human desire and purpose. How does the particular significance of ecosystem services smoothly integrate into the arena of law? As examined by *James Salzman*, scientists must formulate more credible documentation of smaller-scale specificities that cause a reduction in delivery of environmental services and any particularized injury. Only then can legal challenge move productively against environmentally harmful actions. But how

does the economist visualize incremental values of ecosystem services? Individual preference, voting stratagem, as well as a contingent valuation that considers opportunity cost, all of these build into any ecosystem measure as put forth by *David Starrett*.

Beyond market value, forest ecoservices fundamentally contribute to human well-being. However, what quantifiable value do forest resources hold for us? *Bengt Kriström* discusses valuation studies and their methods of measurement especially for non-market forest output. She then examines the usefulness of this valuation toward forest husbandry. Ecological externalities are factors that originate in one ecosystem but contribute to another; they equally have economic application. *Edward Barbier* explores three examples for watershed externalities, with case studies from upstream water diversion in a northern Nigerian river basin; watershed deterioration for upper Javan farmlands; as well as timber production and water yields along the upper catchment of the Colorado River basin in the United States. Markets drive resource use; markets place explicit value on discrete environmental services. *John Gowdy* cautions that market exchange value represents only a partial component of environmental valuation, which is itself, a broader, and perhaps not resolvable, measure of biosphere resources. Modern global economies benefit from the recent revolution in knowledge systems, and there is a growing market for knowledge worldwide. This resurgence occurs in the context of increasing environmental perturbations worldwide, and with patterns of resource utilization that may plunge below replaceable levels, especially in developing nations.

Graciela Chichilnisky asserts that we move toward a more conservative resource use if biodiversity can be intrinsically recognized as intellectual property, e.g., its genetic content and our utilization of this information. As intellectual property rights, this then unexpectedly relates to more familiar property rights, and both may guide us more accurately toward globally sustainable societies.

Resource conservation may itself be financed from the economic valuing of ecosystem services. Management of more than 10% of land area for its watershed value for municipal water supplies is globally possible now, according to *Walter Reid*. Likewise seen as plausible is management toward carbon sequestration in tropical forests at coverages of 5–13% of land areas. However, other ecosystem services (pest control, flood protection) seem less amenable to financial support or evaluation for the resource's conservation. Developing countries may often lack expertise in environmental economics. *Partha Dasgupta* presents one model teaching and research program sponsored by the Beijer Institute as a successful antecedent to a proposed and more permanent institute in the South. *Iddo Wernick* tracks material and energy flows in American industrial applications inspired by their similar paths and interactions in natural ecosystems. In a successful analogy, industrial ecology develops a prognosis for resource movements at a level of numerical precision that is now applicable to policy-making and economic frameworks. Market valuation of environmental resources has not historically led to the conservation of these associated resources, according to *Geoffrey Heal*. Sustainability relates to future perceptions of value, appreciation of benefit, as well as incentive. Economic incentives

will exist if natural processes can be seen as global public goods, e.g., carbon sequestration, watershed management, bioprospecting, even ecotourism. Conservation of a public good (water, wildlife) may thus arise from goods or services that can be commercially valued, enhancing incentive.

What effect does globalization have on Latin American economies? Technological transfer and the Knowledge Revolution clearly have impacts, but the ecological future of South America is equally tied to social issues. Socioeconomic strategies must weigh the scientific and social needs, and in both a Northern and a Southern context. This lies concomitant to resource conservation, as put forth by *Gilberto Gallopín* and *Graciela Chichilnisky. Fernando Gabriel Irrera, Mariano Javier Oneto*, and *Jorge Rabinovich* examine the question further, offering examples from Argentinean legislation detailing specific property rights with respect to natural resources. What legal constraints may exist against resource sustainability in developing countries? Historically, accrual costs to natural systems have not been figured into perceived economic gains. Concessions to private usage do not conflict or pertain to domain over public goods such as ecoresource. Private use of public natural resources can convey a sense of property yet stop short of any actual ownership of the resource itself.

Carlos Galperín and *Jorge Rabinovich* consider alternative policy instruments for developing countries, covering the potential environmental impact of these economic and regulatory legislative instruments. Accompanying this, they conjecture that the Knowledge Revolution will stimulate innovation and economic growth but at what cost to Earth's natural resources and their sustainability?

Finally, in a thorough review, *Jorge Rabinovich* summarizes bionomic repercussions from the perspective of the South. What are the biological and socioeconomic effects of a knowledge-intensive global economy on developing countries, and how will these shape their future direction and response? This appropriately concludes the volume in that one of the chief values of this interdisciplinary conference was to establish a careful dialogue on human impact on our biosphere and then to frame future questions in a heightened specific context.

Ecosystem Services: Characterization of the Function and Principal Threats

How Changes in Biodiversity May Affect the Provision of Ecosystem Services

Shahid Naeem
Department of Zoology
University of Washington
24 Kincaid Hall
Box 351800
Seattle, Washington 98195, U.S.A.
e-mail: naeem@u.washington.edu

Abstract

The collective biogeochemical activities of the biosphere's ecosystems results in the transformation and movement of chemicals among the lithosphere, hydrosphere, and atmosphere. Although largely invisible or seemingly minor to humans, these biogeochemical processes are responsible for the movement of gigatons of elements, chemicals, and materials and are ultimately responsible for regulating the earth's surface conditions. Lose the biota, and the earth's surface becomes a sterile geochemical system whose equilibrium state is likely to be somewhere between that of Venus and Mars; one that is completely inhospitable to life.

The obvious dependency of humans on biogeochemical processes makes biogeochemically defined ecosystem services among the most readily quantified of all ecosystem services and arguably the most valuable. On a global scale, such services include the regulation of greenhouse gasses and local and global climate. On regional and local scales, these ecosystem services include the retention of nutrients, water, the cycling of elements and chemicals, decomposition and degradation of pollutants, and the production of food, fuels, lumber, and other biological products.

One of the most dramatic global changes the earth is experiencing is anthropogenic biodiversity loss, but the role of this diversity in overall ecosystem functioning is poorly understood. Although it is relatively clear that complete loss of the biota would mean a complete transformation of the Earth's surface to a sterile planet, the biogeochemical consequences of retaining the Earth's biosphere but substantially reducing its biotic diversity is unclear. Worded differently, if a prairie grass-

land consisting of 200 species of grasses and forbs per hectare were changed to one with two plant species per hectare (e.g., corn and soybean), or if a tropical rainforest consisting of 300 species of trees per hectare were changed to one with a single species of tree (e.g., pine), how biogeochemical processes of these ecosystems would respond is difficult to predict.

Recently, a number of ecological studies have examined the ecosystem consequences of declining biodiversity. Current experimental and theoretical studies demonstrate that there are a number of potentially serious consequences of biodiversity loss. In some instances, the actual level of service provided by an ecosystem will change if biodiversity is lost. Perhaps even more important, the predictability or reliability of ecosystems as providers of biogeochemical services may also be challenged by biodiversity loss. This recent research is limited in scope and its findings subject to different interpretations, but its implications are clear: reduce biodiversity to very low levels, typical of many managed ecosystems, and ecosystem processes change. Thus, humans, who have now taken the helm of the biosphere, may find themselves managing a planet whose performance is both different and less reliable than the one they have been living on for the last 3 million years.

Introduction

Understanding the implications of ecological change for human welfare can be tackled in a variety of ways. From an ecologist's perspective, the first order of business is to delineate how ecological processes (e.g., biological processes such as growth, reproduction, dispersal, and feeding of organisms within a habitat) govern environmental processes (e.g., physical and chemical processes within a habitat). This step necessarily requires excluding humans from the picture in order to establish how nature functions in the absence of human impacts. The second step is to determine how humans have or will alter these "natural" ecological processes and thereby possibly alter environmental conditions. The third step then is to determine how such human alterations of nature may affect human welfare and, depending on one's ethical foundation, the welfare of other organisms. At this point economists, sociologists, environmentalists, philosophers, or others generally take the baton from ecologists and carry the analyses forward. The purpose of this symposium was to make this third step a more integrated process rather than a baton-passing exercise. Ecologists should provide perspectives on how organisms affect the environment, while economists and other social scientists should provide perspectives on how the environment affects human welfare and human economies. A synthesis then emerges from such an exchange.

This chapter provides an ecologist's perspective on one of the most critical issues concerning human impacts on the environment and the potential consequences

of such impacts to human welfare. Specifically, this chapter concerns the environmental consequences of *biodiversity loss* (the local or global extinction of species). This may seem like a fairly restricted focus, but actually, the problem of biodiversity loss is central to understanding how ecosystems, the biota, the environment, and human welfare are intertwined.

The problem of biodiversity loss is enormous, but the ecological and economic consequences of this loss have been barely examined. Contemporary reductions in biodiversity are unprecedented, far exceeding any of the known mass extinctions earth has experienced in its 4.5-billion-year history (Horace, 1985; Wilson, 1988; Soulé, 1991; Pimm et al., 1995; Reaka-Kudla et al., 1997; Stork, 1997). For ecologists, this loss has mainly motivated studies examining how this loss might be stemmed rather than asking what the environmental consequences of such loss might be. Similarly, this loss has attracted the attention of economists (Perrings et al., 1995), but the focus has been on loss of goods (e.g., foods, furs, ornaments, lumber), genetic resources (e.g., gene products for drugs), or recreation value (e.g., trophy hunting, zoos, parks, tourism), rather than the value of biodiversity for its environmental services.

In truth, lack of attention to this matter is most likely because the environmental role of biodiversity has been difficult to assess. Quantifying biodiversity is difficult, and uncertainty surrounds estimates of taxonomic diversity (estimates range from 10 to 100 million species (Lawton & May, 1994; May, 1995)). But even if we adopt the conservative convention of using 12.5 million species as an operational estimate of species diversity (Groombridge, 1992), there is no question that an extraordinary amount of biological diversity exists within the biosphere, the role of which is hardly understood. How does one begin to examine a system with 12.5 million unique parts? One might begin by examining local systems rather than the globe, but biodiversity even on a local scale is staggering. A single gram of soil can contain 10^4 species of microorganisms (Klug & Tiedge, 1994), a prairie grassland can contain nearly 200 species of herbaceous plants (Leach & Givnish, 1996), a Peruvian tropical canopy can contain 3429 species of beetles (Erwin, 1997), and temperate soils can contain around 150 microarthropod species (Usher et al., 1979). Almost anytime and anywhere that biotic inventories have been conducted, diversity generally is much higher than what existing records would lead one to believe. As this chapter will outline, although it is clear that *some* portion of this diversity is absolutely essential for the maintenance of environmental functioning, until recently it has been relatively unclear whether *all* or even a large portion of this diversity is necessary for a healthy environment.

In this chapter, I will review the environmental consequences of biodiversity loss by first providing an ecological overview of the relationship between organisms and their physical and chemical environment. I will then review the current state of our scientific understanding of the problem. Of course, I should point out at the outset that the significance of biodiversity loss extends beyond alterations of biogeochemically defined ecosystem services and includes many market and

non-market values (Kunin & Lawton, 1996; Perlman & Adelson, 1997), but these other issues are outside the scope of this chapter.

The Biota and the Environment

The metabolic activities of organisms within ecosystems primarily involves consuming energy to transform and concentrate chemical elements and compounds. That is, the consumption of energy by organisms is used to acquire and locally concentrate materials from the environment that are necessary for growth and reproduction. The end result of these activities is the growth of populations, though numerous sources of mortality generally keep most populations in check. Whether a bacterium, whale, yeast, or a large tree, the biochemical mechanisms for accomplishing these metabolic activities are remarkably similar. The principle elements being moved, whether in pure form or as compounds, consist of C, H, N, O, P, and S with K, Ca, Mg, and Na also playing important roles. Several other elements such as Fe, Mo, Zn, and Cu are additionally critical for many biochemical pathways, but much smaller quantities of these elements are moved by organisms. This biologically mediated movement of elements and chemicals among the atmosphere, hydrosphere, and lithosphere is known as *biogeochemistry*. Here, I review biogeochemistry briefly. Technical reviews of biogeochemistry (Butcher et al., 1992; Schlesinger, 1997) and an excellent, more general text (Smil, 1997) are available for the interested reader.

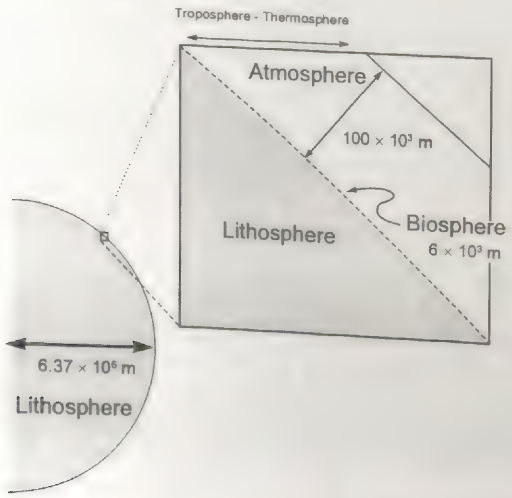


Figure 1. The biosphere in relation to the lithosphere and atmosphere drawn approximately to scale. This figure illustrates the extraordinarily "thin" nature of the biosphere, which makes its value somewhat difficult to perceive by humans because it seems so insignificant in size by comparison to the other spheres. Note that although "thickness" undoubtedly varies, at this scale such variation would not be visible. Clearly deserts do not have as much biomass per unit area as most tropical rainforests.

Biogeochemistry

Because the atmosphere, lithosphere, and hydrosphere are enormous in size by comparison to the biosphere, it is often difficult to appreciate the role of the biota in determining the conditions of the earth's surface. If the earth were about the size of a basketball, the biosphere would be like a sheet of thin paper wrapped around the globe (Fig. 1) (Lovelock, 1979). This "thinness" is due simply to the harsh physical conditions that occur at high altitudes, such as low temperature (-60°C at the top of the Troposphere), little water vapor, low concentrations of oxygen, high UV radiation, and other physical changes that occur as one moves away from the planet's surface. Areas of this paper-thin sphere, such as cities, deserts, and arctic environments, would be very "thin" in the sense that the quantity of biomass per unit area would be very low compared with other regions, such as coastal habitats and forests, where the quantity of biomass per unit area might be very high. The biosphere might be considered "thick" in these regions compared with deserts or cities. Over the course of a year, in spite of the thinness and unevenness of the biosphere, gigatons (10^{15} g) of materials are moved among the four spheres (Fig. 2). For example, over 140×10^{14} g N yr^{-1}, 60×10^{12} g P yr^{-1}, 20×10^{12} g S yr^{-1}, and 120×10^{15} g C yr^{-1} are moved by biologically mediated processes each year (Schlesinger, 1997). This movement of material is fueled by energy acquired by photosynthetic organisms (primarily algae in the ocean and plants on land), which yields a global net primary production of about 110×10^{15} g C yr^{-1} (Schlesinger, 1997). The energy and carbon acquired by these organisms is of course used for their vegetative and reproductive growth, but non-photosynthetic organisms can obtain their energy and carbon needs by consuming either living or dead photosynthetic organisms (see discussion below). Without production, there would be no biogeochemical activity. For this reason, both on a global level and on local levels, production or productivity is often the most intensely studied process.

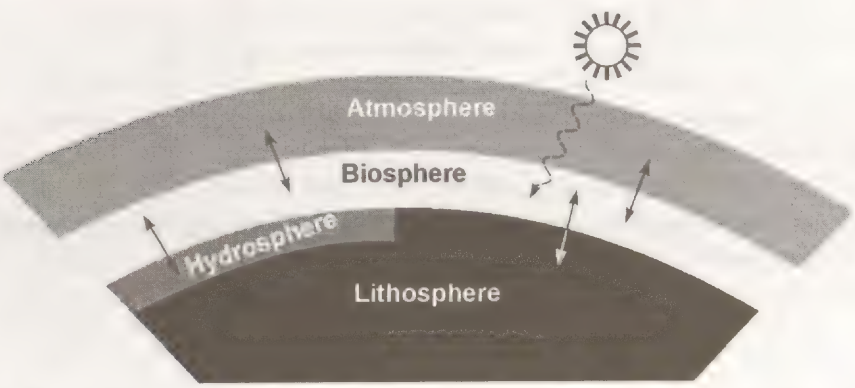


Figure 2. Biogeochemical exchange among the earth's spheres (not drawn to scale—see Fig. 1). The biosphere may be thought of as a thin tissue that moves material among the earth's spheres. The bulk of these biogeochemical exchanges are accomplished by the collective metabolic activities of organisms fueled by solar energy (wavy line entering biosphere).

Table 1. The role of the biosphere in governing surface conditions of the planet. Comparison of present conditions (BIOTIC EARTH) with potential conditions of earth without a biota (ABIOTIC EARTH). Conditions of Venus and Mars are provided to show that abiotic earth might appear less anomalous as part of the solar system in the absence of the biosphere. (After Lovelock, 1979.)

GAS	PLANET			
	VENUS	ABIOTIC EARTH	MARS	BIOTIC EARTH (PRESENT)
CO_2	98%	98%	95%	0.03%
N_2	1.9%	1.9%	2.7%	79%
O_2	trace	trace	0.13%	21%
Temp. (°C) (surface)	477	290±50	-53	13
Atmospheric pressure	90	60	0.064	1.0

Because the climatic, hydric, and edaphic (soil) conditions of the earth's surface are determined by the chemical composition (e.g., percent O_2, CO_2, N_2, and H_2O), physical properties (e.g., clouds, reflectance, or spectral absorption) and structural properties (e.g., stratification) of the atmosphere, lithosphere, and hydrosphere, the biologically mediated movement of such tremendous quantities of elements and chemicals among the earth's spheres clearly shows that the biosphere is critical in determining the surface conditions of the earth. As shown in Table 1, a simulation study shows that without the biosphere, the earth becomes an uninhabitable planet for most of the biota, including humans. This prediction should not be surprising given that photosynthesis alone was responsible for producing the enormous quantities of oxygen that would first chemically bind to vast quantities of reactive elements such as iron and sulfur on the earth's surface and then begin to accumulate to its present levels of 20% of the atmosphere as far back as 430 million years ago (Schlesinger, 1997).

Humans are altering the structure and composition of the biosphere. Two sorts of anthropogenic alterations of the earth's biota are of concern. The first is a physical restructuring of the earth's surface involving changes in landscape and land-use patterns. These include habitat fragmentation, alterations of hydrology, and urbanization. The second is a change of biotic structure of the ecosystems found on the earth's surface. These changes include both direct modifications by transforming unmanaged ecosystems into managed ecosystems, such as turning grasslands and forests into agroecosystems (Imhoff, 1994), which are invariably low in biodiversity and consist of species quite different from the former tenants, and indirect modifications such as alterations of climate that lead to biodiversity loss (Peters et al., 1992; Kareiva et al., 1993) or the introduction of "exotics" into ecosystems or "biological invasions" (Elton, 1958; Drake et al., 1989; Vitousek, 1990; Cronk & Fuller, 1995). Most of these changes result in the local or global extinction of species (Ehrlich & Ehrlich, 1981; Raup, 1991; Lawton & May, 1994; Sisk et al., 1994; Tilman et al., 1994; Hughes et al., 1997; Tilman et al., 1997d).

Although it is clear that our biota's biogeochemical activities are critical to local, regional, and global climatic, hydrologic, and edaphic conditions, as mentioned

earlier, it is not at all clear that the large numbers of species frequently involved in these processes are necessary. Therefore, it is not immediately clear that human alterations of the biotic composition and structure of the biosphere may have environmental implications. For example, a two-species corn-soybean rotation farm may have replaced a previously diverse biological community such as a 200-species prairie, but both the farm and the prairie it replaced contribute to biogeochemical processes in much the same way. Another example might be the replacement of a tropical forest of 300 species of trees with a monoculture of pine. The monoculture may still be a "rainforest" ecosystem in the sense that the rainfall, soil, and biological activities of the organisms in the habitat are more or less the same, but is this depauperate system truly the same biogeochemical system as the species-rich system it replaced?

A number of recent ecological analyses have examined how biodiversity may or may not affect ecosystem services (Schulze & Mooney, 1993; Vitousek & Hooper, 1993; Collins & Benning, 1996; Johnson et al., 1996; Naeem et al., 1995a & c; Chapin et al., 1997). A rich diversity of hypothetical relationships between biodiversity and ecosystem functioning have emerged, which are illustrated in Figure 3. If we consider that ecosystem functioning (primarily associated with biogeochemical processes) is related to environmental functioning (primarily associated with physical and chemical conditions), then Figure 3 also illustrates the wide variety of ways the environment is related to biodiversity.

If Figure 3 appears to represent a jumble, that is precisely the image that it is meant to convey. Such a variety of hypotheses, as in any science, attests to the

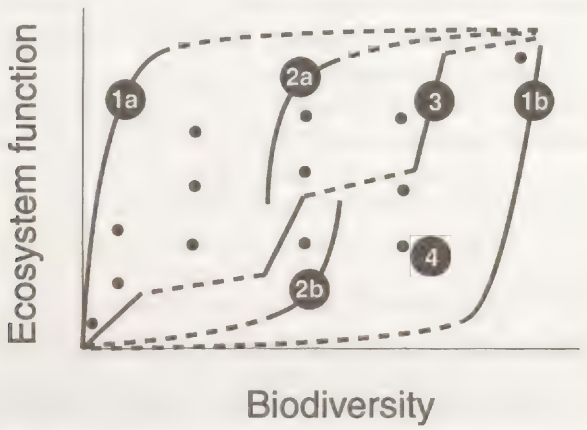


Figure 3. Hypothetical relationships between biodiversity and ecosystem functioning. Ecosystem function can be any biogeochemical process or ecosystem attribute such as stability, nutrient cycling, or production. Biodiversity may be measured as species richness, complexity, or any other index of the biota within an ecosystem. Moving left in the diagram predicts ecosystem response to variation in biodiversity as biodiversity loss and homogenization occur. —1a & b, compensatory/keystone hypothesis (Sala et al., 1996); —2a & b, nonlinear hypothesis (Carpenter, 1996); —3, rivet hypothesis (Ehrlich & Ehrlich, 1981); —4, the idiosyncratic hypothesis (Lawton & Jones, 1993).

youth of the science and provides the foundation from which theoretical and empirical studies begin. Much of the empirical evaluations of these hypotheses are recent, but, as described in the next section, two possible ecosystem consequences of biodiversity loss have emerged from this research. First, levels or the magnitudes of rates of biogeochemical processes may be affected by biodiversity loss. Second, and what I will argue is perhaps the most important, the reliability or predictability of ecosystem services may be affected by biodiversity loss. The next section reviews the evidence for these ecosystem consequences of biodiversity loss.

Biogeochemically Defined Ecosystem Services

The most fundamental level at which the biota's environmental role can be appreciated is at the level of the ecosystem. An ecosystem can be any system in which an assemblage of species (*community*) uses local energy and material resources. An aquarium is as much an ecosystem as a tropical rainforest or farm. In reality, such boundaries are artificial because no ecosystem is truly isolated from another since, at some level, all ecosystems exchange materials with one another. As discussed above, this linkage of ecosystems occurs through exchange of gaseous materials through a shared atmosphere (e.g., O_2 or CO_2) or through long-term cycling processes involving the movement of nutrients or materials through the hydrosphere and lithosphere (e.g., P, N, or K). For practical purposes, however, an ecosystem is defined as a reasonably bounded assemblage of populations whose biological activities contribute to the flow of energy and materials through the system. How an ecosystem's biota and variation in the composition and structure of this biota affect such flows is the key to understanding the role of biodiversity in governing local and global environmental conditions.

Ecosystem services are numerous (de Groot, 1992; Daily, 1997; Daily et al., 1997), but only some of these are associated with the biotic transformation and movement of materials, or biogeochemically defined ecosystem processes. This section examines the nature of such processes and their relationship to biodiversity to provide an overview of the foundation for the empirical and theoretical research reviewed in the next section.

Biodiversity: Taxonomic versus functional diversity

From an ecosystem perspective, taxonomic or genetic diversity as estimated by the number of species or *species richness*, the conventional metric of biodiversity, is not as important as *functional group richness* (functional diversity) or classification by biogeochemical activities. Estimating functional diversity requires classifying species by their functional attributes, a practice long exercised by ecologists (Raunkier, 1934; Root, 1967; Körner, 1993; Gitay et al., 1996). Such practices stem from the recognition that many species may be treated as operationally redundant with respect to ecological or ecosystem processes even if they are phylogenetically or

genetically distinct. Such *ecological redundancy*, in which one species is effectively substitutable by another, is widely recognized as a common feature of ecosystems (Walker, 1992; Lawton & Brown, 1993; Walker, 1995; Gitay et al., 1996). "Redundancy," although referring primarily to ecological equivalency, carries the unfortunate connotation of being "expendable," which is not at all what ecologists mean by ecological redundancy (Walker, 1995; Naeem, 1998). This unfortunate confusion has generated some misunderstanding in the use of the term.

An important, trophically defined functional division among organisms is whether their biochemical capabilities permit them to be *autotrophs*, or "self feeders" like photosynthetic plants or chemosynthetic bacteria. Such autotrophs are capable of acquiring energy from non-biological sources (e.g., light energy via photosynthesis or chemical energy via chemosynthesis). As such, autotrophs are often referred to as producers. All other organisms are known as *heterotrophs*, individuals that must obtain their energy by consuming others. Heterotrophs can extract energy from autotrophs by consuming living autotrophs (e.g., herbivores), dead organisms (e.g., decomposers), or consuming other consumers (e.g., predators, parasites, omnivores).

Species may be classified by their biotic interactions with other species. Biotic interactions link species within a community either by the way they share resources or how nutrients and energy flow through chains or webs of interacting consumer species. Such interactions include competition, predation, mutualism, parasitism, commensalism, and many others. The pattern of such linkages among species in a community is known as an ecosystem's *community structure*. If we ignore all biotic interactions except feeding interactions, the pattern of trophic or feeding linkages among species is known as an ecosystem's *trophic* or *food-web structure*. In general, ecologists assume that competition for resources is greater within a trophic group than among trophic groups. Similarly, the argument can be made that redundancy is greater within functional groups than among functional groups.

Ecosystem processes

Ecosystem processes are ultimately the processes which provide the goods and services that are the focus of much of environmental economics. From an ecological perspective, however, ecosystem processes have a specific meaning and are readily quantifiable and analyzed by empirical and theoretical methods. There are seven basic ecosystem processes (Fig. 4) that categorize the major flows of energy and materials among producers, consumers, and the abiotic environment. These are (1) assimilation of nutrients by autotrophs, (2) energy acquisition by autotrophs such as photosynthesis or chemosynthesis, (3) respiration or the reverse of energy acquisition, (4) death, senescence, or contributions to the pool of dead organic matter (from autotrophs and heterotrophs), (5) decomposition or the transformation of organic nutrients in dead material to inorganic forms, (6) immobilization or the accumulation of inorganic minerals by microbes in the decomposer subsystems, and (7) nutrient mineralization or return of nutrients to inorganic pools. Ecologists have literally hundreds of ways of quantifying these processes as rates, such as rates of

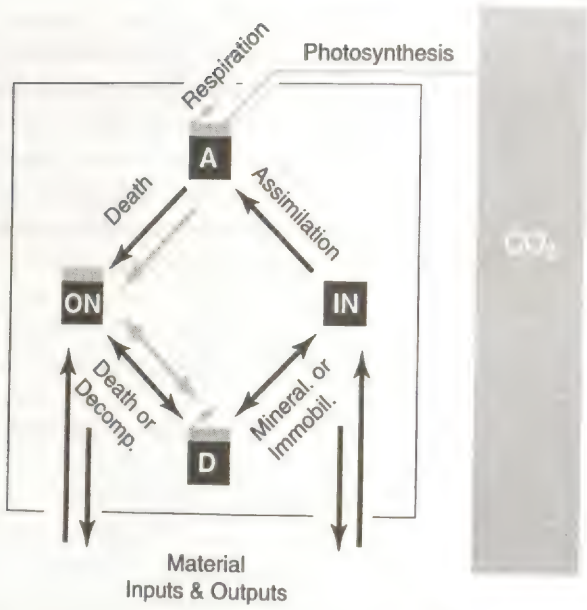


Figure 4. The basic ecosystem processes and compartments. Autotrophs (A) acquire energy through photosynthesis, which is stored in carbon bonds in which the carbon is derived from the atmospheric pool of CO_2. Carbon is shown as gray portions of the boxes representing autotrophs, decomposers (D), and the pool of organic nutrients (ON). Loss of carbon by respiration is shown as small, gray arrows. Inorganic nutrients are represented by a separate pool (IN). Note that most ecosystems exchange material with other ecosystems through inputs and outputs or through photosynthesis and respiration. Not shown is the fact that organisms may also immigrate and emigrate and that additional consumer species are likely to be present.

decomposition (determined by mass loss of dead organic material), rates of photosynthesis (measured as net CO_2 flux by such devices as infrared gas analyzers), or rates of nutrient cycling (measured by stable isotope analyses or fluxes of radioactively labeled nutrients). There are many other ecosystem processes, but all are related in one way or another to these basic processes.

Note that biodiversity *per se* is not a necessary part of understanding ecosystem processes. An ecosystem need only contain one species of producer and one species of decomposer for it to be a viable ecosystem performing all seven of the major ecosystem processes or functions. But ecosystems are generally extraordinarily rich in species and look more like the complex structure portrayed in Figure 5.

These processes are perhaps alien to those who typically think of ecosystem services as climate regulation, mitigation of floods, purification of air, or other widely discussed "services." Yet, these processes lie at the root of all ecosystem services. For example, climate regulation occurs in part because photosynthesis and respiration annually move gigatons of carbon into and out of the atmosphere, thereby determining the concentration of such greenhouse gasses as CO_2. Mitigation of floods might occur because autotrophic organisms, such as vegetation along ripar-

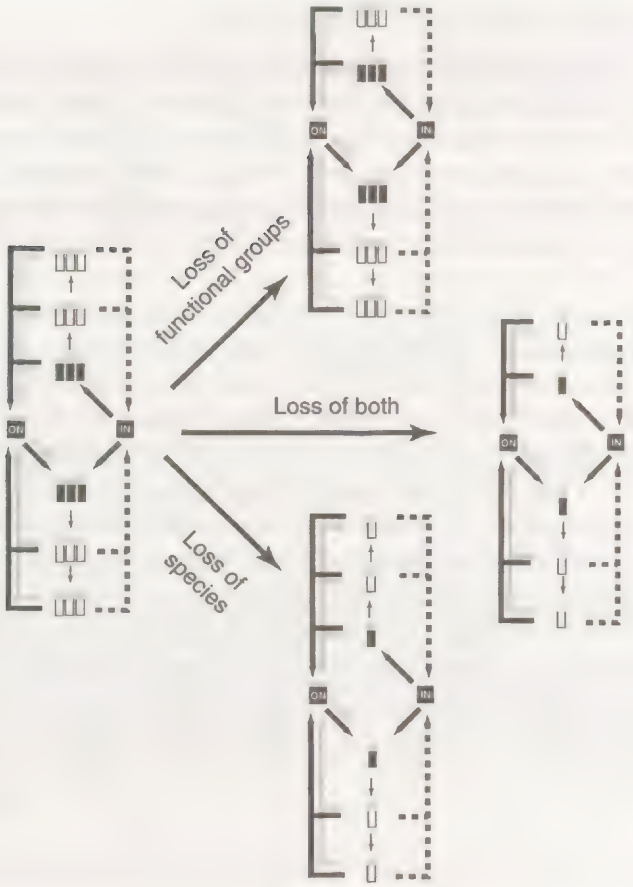


Figure 5. Ecosystem structure and biodiversity loss. Following Figure 4, the core of an ecosystem is shown here as black boxes. Most ecosystems, however, contain many species of autotrophs and decomposers and many species of consumers (open boxes), which feed (thin, central arrows) on either autotrophs or decomposers or both (not shown). All these organisms contribute to the organic nutrient pool (ON) via death (solid lines) and to the inorganic nutrient pools (IN) via mineralization (dashed lines). Defining functional groups as trophic groups, biodiversity loss may involve the loss of functional groups, loss of species within functional groups, or the loss of both. These three kinds of changes are likely to have different consequences to overall levels of ecosystem processes, as well as dynamic properties of both populations and ecosystem processes.

ian zones and watersheds, anchor the soil and move water from the soil into the atmosphere through photosynthesis and respiration, which is accompanied by evapotranspiration. Purification often involves a myriad of biochemically diverse decomposer species degrading and removing organic compounds from air or water. It is worth mentioning again that the starting point of all ecosystem processes and hence all ecosystem services is energy acquisition (usually photosynthesis), which is why production is so widely studied among ecosystem processes.

Biodiversity and ecosystem functioning

"Functioning" is a tricky word because it carries with it a sense of purpose or conscious design. In its most basic sense, however, the word simply refers to activity of a system. A functioning system is one showing activity typical of that system, where "typical" refers to long-term averages and variances in functioning. When that activity changes such that the system shows *levels* of activity that are significantly different from long-term trends, then the system may be declared not functioning. This does not mean that the ecosystem has collapsed or no longer exists, but from a human perspective, which is the relevant perspective for assessing ecosystem services, it is no longer functioning. For example, if a clear lake that once provided drinking water, fish, and recreation becomes eutrophic because of pollution, the fact that the lake has become turbid with dense populations of algae, has low water quality, is incapable of supporting fish, and has lost its value for recreation, means that it is no longer functioning. However, if our desire changes to one in which we wish to have a lake that sequesters carbon from the atmosphere rather than provide drinking water, fish, and recreation, then we have redefined functioning for this system and as a lake filled with active algae, it is functioning in this new sense of the word.

Patterns in the temporal *dynamics* of ecosystem functioning are also important. Levels of functioning are important, but changes in the dynamics of ecosystem functioning can be equally important. Changes in dynamics can include changes in variance, changes in recovery times from a disturbance (resilience), or changes in sensitivity to disturbance (resistance). For example, an ecosystem can continue to show the same mean level of production, but if its variance in production doubled, or after a fire or drought the system took twice as long to recover (lowered resilience), or lost twice as much biomass to the fire than had ever been observed before (lowered resistance), then the functioning of the ecosystem has changed just as significantly as if its level of production had been halved.

Biodiversity and Ecosystem Functioning: The Evidence

Although a substantial amount is known about the ecology of communities and ecosystems, little information has been available on the relationship between biodiversity and ecosystem functioning (Schulze & Mooney, 1993; Orians, 1997; Chapin et al., 1998). Population ecologists have used both community structure (MacArthur, 1955; Levin, 1970; May, 1974; Lawler & Morin, 1993; Haydon, 1994) and food-web structure (Hairston et al., 1960; Pimm & Lawton, 1977; Pimm, 1982; Cohen, 1991; Polis & Winemiller, 1996) to understand the dynamics and stability of ecological communities. These studies have shown quite clearly that changes in structure affect the dynamics of populations within an ecosystem. Because the dynamics of nutrient and energy flow are affected by the dynamics of populations, it is clear that changes in structure affect ecosystem processes (De Angelis, 1992; Loreau, 1995). Ecologists, however, have disagreed, and there has been little consensus among ecologists that diversity (taken here to be synonymous with "complexity") and stability are related

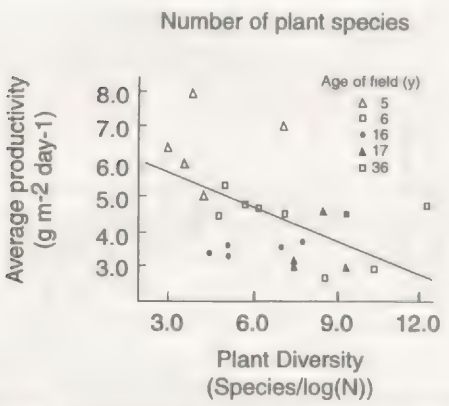


Figure 6. Observational study of production in New York old field grasslands. (Adapted from McNaughton, 1993.) Note that fields of similar ages are closely associated with one another, thereby making it difficult to know if the association between production is driven by differences in age among the fields or differences in plant species richness.

in any simple way or that diversity and overall levels of ecosystem functioning are related. For this reason, a number of theoretical and empirical tests of the ideas shown in Figure 3 have been conducted over the last five years.

Like with most ecological problems, three approaches have been taken; (1) observational, (2) experimental, and (3) theoretical. Scientific evidence in any discipline is the intersection of theory, experiment, and observation. When information from these three lines of investigation agree, then progress has been made. Here, I review current observational, experimental, and theoretical evidence that biodiversity and ecosystem functioning are related.

Observational studies

This first approach one might take would be to correlate measures of biodiversity with measures of ecosystem functioning based on observations of natural systems, but such an approach by itself provides information that is difficult to interpret in the absence of experimentation. The relationship between biodiversity and production as determined from observational studies appears to either be weak or inverse (McNaughton, 1993; Bulla, 1996). Such studies, however, require locating regions that differ in biodiversity and examining how variation in diversity covaries with ecosystem processes. In such instances it is generally difficult or impossible to know if there is some unmeasured factor that affects both diversity and ecosystem functioning, and the observed correlation between diversity and functioning is spurious. Figure 6, for example, shows an apparent strong, inverse correlation between plant diversity and production in a series of old fields in New York. Note, however, that high-diversity communities are also more mature communities. It is therefore not possible to separate whether successional or aging processes in an ecosystem lead to the lowered production or whether higher biodiversity is responsible for the lowered levels of production. Similarly, Wardle and colleagues (Wardle et al., 1997b)

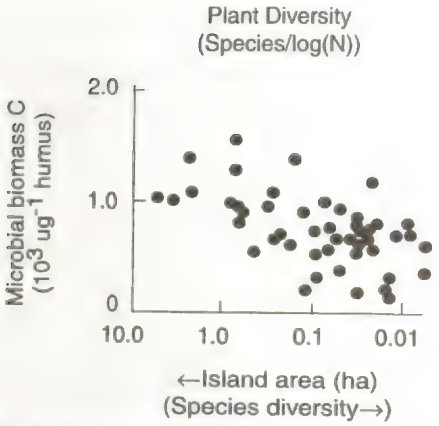


Figure 7. Observational study of a below-ground ecosystem process in relation to plant biodiversity. In this system, island size is inversely correlated with plant diversity due to higher frequencies of lightning strikes and fires on larger islands. The bottom axis is therefore drawn to show both island size and plant species richness. Islands are boreal forested islands from a Swedish archipelago. (Adapted from Wardle et al., 1997b.)

argued for inverse associations between diversity and several below-ground ecosystem processes. Figure 7 shows one such inverse association between microbial biomass and plant diversity. Again, the difficulty with interpreting these results is that the islands used in Wardle et al.'s study differed in size, fire frequencies, and species diversity, with larger islands having more fires and fewer species. It is therefore impossible to determine if the associations between below-ground ecosystem processes and island size are entirely due to fire frequency (a factor that dramatically alters above- and below-ground communities and ecosystem processes) or the incidental variation in biodiversity. These studies do, however, show that extrinsic factors are frequently associated with diversity, a factor well known among ecologists (Whittaker. 1975; Currie, 1991; Rosenzweig & Abramsky, 1994). That is, they demonstrate that variation in environmental conditions may lead to changes in biodiversity or vice versa.

Observational studies of the roles of individual species have been much more successful at demonstrating the importance of a region's biota to ecosystem functioning. There are many cases in which a single or small group of species can have dramatic impacts on a community or ecosystem. Such species are sometimes termed *keystone species* because of their disproportionate effect on a community or ecosystem structure or processes when compared with other species in the system (Paine, 1966; Mills et al., 1993; Jones et al., 1994; Menge et al.; 1994; Liebold, 1996; Power et al., 1996). For example, a single species of nitrogen-fixing tree, *Myrica faya*, in Hawaii has had significant effects on nitrogen cycling and primary succession (Vitousek, 1990). Moose (*Alces alces*) have also been shown to have strong impacts on biogeochemistry (Pastor et al., 1993). In some cases, entire landscapes may be affected by groups of organisms such as termites (Macrotermitinae) on arid grasslands in Africa (Jones, 1990; McCarthy & Ellery, 1994), beavers (*Castor canadensis*)

in temperate habitats (Naiman et al., 1986; Kronberg, 1993), or large grazers in tundra/steppe ecosystems (Zimov et al., 1995).

Experimental studies

Determining how variation in biodiversity affects ecosystems requires direct manipulations of diversity followed by measuring ecosystem response to such experimentally generated variation. Such experiments have only recently begun, but they have shown some interesting patterns of association between diversity and functioning.

The first such study was conducted in an integrated series of controlled environmental chambers known as the Ecotron (Lawton et al., 1993; Naeem et al., 1994; Naeem et al., 1995c; Lawton, 1996), with a companion experiment conducted in a greenhouse (Naeem et al., 1995b & c; Naeem et al., 1996). This experiment manipulated the number of species within several trophic groups and manipulated plant diversity more thoroughly in the greenhouse experiment. Together these experiments showed that such ecosystem processes as production, nutrient retention, and respiration responded significantly to variation in biodiversity. Ecosystem response varied, however, with some functions showing positive relationships with diversity while others showed inverse or no relationship. The important point was that under completely controlled conditions (i.e., no potential for spurious correlations with covariation in external factors), variation in biodiversity itself was sufficient to significantly alter the biogeochemical behavior of ecosystems. Naeem et al. (1994) argued that the mechanism behind many of these responses was most likely the greater efficiency of light utilization by the more diverse plant communities. Naeem et al. (1996) cautioned, however, that specific relationship between plant diversity and production may be due to the nature of the pool of species from which the replicate communities are drawn. That is, change the composition of the species pool and a different relationship between plant diversity and production could be obtained.

Field experiments with plants, both large scale (Tilman et al., 1996; Tilman et al., 1997a) and small scale (Hooper & Vitousek, 1997, 1998), have shown that variation in plant diversity, either in terms of species richness or functional group richness, significantly affected above-ground biomass and below-ground N pools (Fig. 8). Tilman et al. (1996) argued, based on soil nitrogen responses to variation in plant diversity, that the mechanism for this positive relationship between diversity and production was the greater efficiency of nutrient use by more diverse communities. Although a positive association between production and diversity was noted, the association was asymptotic (Fig. 8) as observed in the Ecotron studies.

The relationship between diversity and production may also be perceived of as what Pacala has informally called the *sampling effect*. This effect is a positive association between production and plant diversity that occurs because more diverse communities have a higher probability of including more productive species. Because some ecologists feel that statistical properties are not legitimate properties of biodiversity, the sampling effect has also been labeled the "hidden treatment effect" (Huston, 1997). This unfortunate terminology erroneously implies that

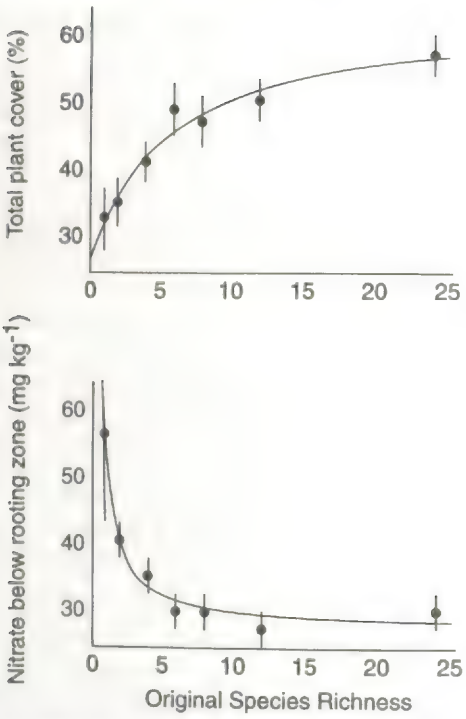


Figure 8. Experimental study of prairie grassland production in relation to plant species richness. Experimental grassland plots are from Cedar Creek, Anoka Co., Minnesota. In this experimental system, all plots are the same size and age and experience the same climatic conditions. (Adapted from Tilman et al., 1996.)

statistical values of diversity are unimportant when, in fact, they could be one of the strongest benefits of having high regional diversity. That is, higher regional diversity increases the probability of local systems that draw from this pool (through immigration and emigration) of having highly productive plant assemblages.

Roy and colleagues showed that mature plant communities, removed from the field to controlled environmental chambers and planted with new vegetation, did not show the same response in production observed in the relatively immature experimental systems described above (Chapin et al., 1998). This experiment showed that microbial biomass increased with plant diversity in three treatments (Fig. 9). Thus, young and old communities may show different responses to changes in plant diversity, as observed in Figure 6, pointing to why observational and experimental studies may not agree with one another, but together provide a more clear picture of the actual relationship.

Aside from the Ecotron study, only two other studies have conducted multitrophic-level manipulations of biodiversity. Both of these experiments used microcosms, varying the diversity of microbes (protists, bacteria, and algae) or other small invertebrates in large numbers of replicate communities containing several trophic levels (producers, consumers, and decomposers) (McGrady-Steed et al., 1997;

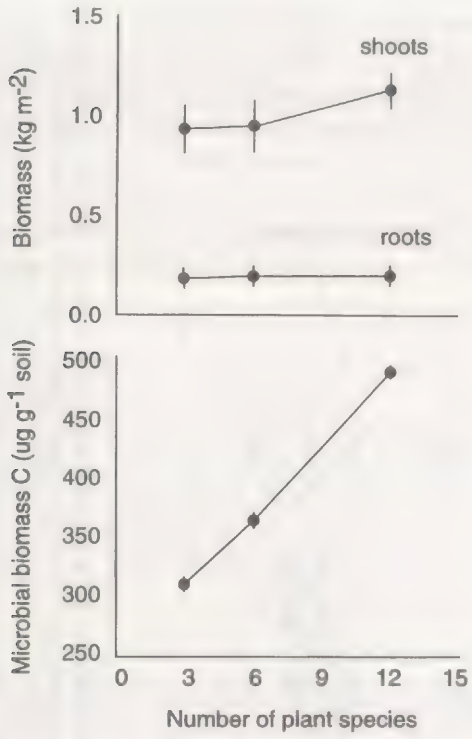


Figure 9. Experimental study of plant diversity and microbial biomass in mature meadow communities. Experimental communities were taken as monoliths from the field, placed in controlled environmental chambers within greenhouses, and plant diversity was experimentally manipulated by planting different numbers of species of native plants. Experimental monoliths were from old field Mediterranean herbaceous plant communities. (Adapted from Chapin et al., 1998.)

Naeem & Li, 1997). Because of the highly dynamic nature of these model systems, local but random extinction, and large number of replicates, variation in the performance of these systems could be examined to test how predictability or reliability of ecosystem functioning was affected by variation in biodiversity across several trophic levels. In fact, Naeem and Li (1997) showed that the improved reliability of ecosystem functioning due to higher levels of diversity was detectable across variation in both light and nutrients (Fig. 10). "Reliability" and "predictability" refer to probabilities that ecosystems will exhibit predefined levels of functioning over a specific unit of time (Naeem, 1998). These terms are treated in more detail in the next section.

Contemporary theory

A recent set of theoretical papers have attempted to explain empirically derived patterns of association between biodiversity and ecosystem functioning. The approaches vary from traditional use of Lotka-Volterra or resource-based competition models (Tilman et al., 1997b; Hughes & Roughgarden, 1998), to less conventional

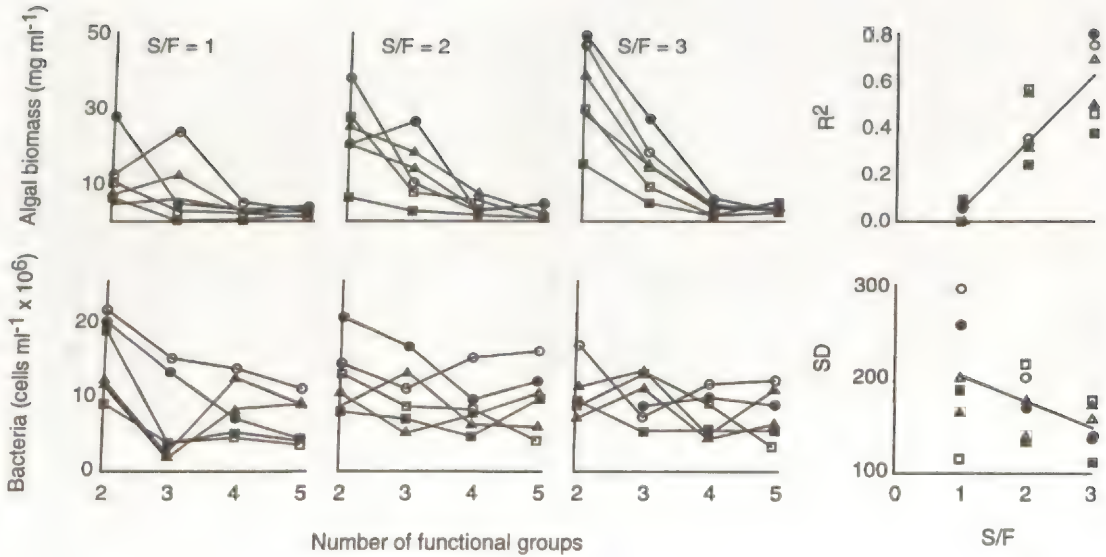


Figure 10. Experimental study of autotrophic and decomposer production in relation to non-decomposer, heterotrophic diversity. Experimental systems were microbial microcosms containing algal and heterotrophic protists plus bacteria serving as decomposers. Both numbers of trophically defined functional groups and numbers of species per functional group (*S*/*F*) were manipulated. Light (closed and open symbols) and nutrients (circles, triangles, and squares) were also experimentally varied two-fold and three-fold, respectively. Right-most plots show the relationship between measures of system predictability, and *S*/*F*. R^2 is a measure of goodness of fit of the linear relationship between ln (algal biomass) and number of functional groups. SD is standard deviation of bacterial biomass. (Adapted from Naeem & Li, 1997.)

uses of ecosystem models (Loreau, 1998), reliability engineering models (Naeem, 1998), and statistical models (Doak et al., 1998).

Two models have focused on levels of ecosystem functioning. Tilman et al. (1997) and Loreau (1998) both used interspecific competition for nutrient resources as the bases for their models and derived relationships between plant (producer) diversity and levels of production (ecosystem functioning). Their results are strikingly similar to what has been observed in experimental tests (compare Fig. 11 with Fig. 8), both showing asymptotic relationships between levels of ecosystem functioning and species diversity.

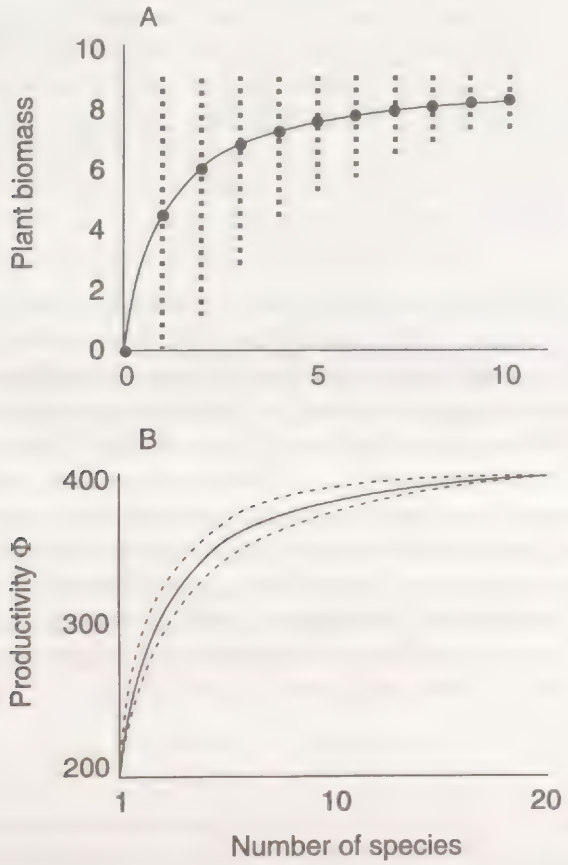


Figure 11. Results from two theoretical models predicting ecosystem production as a function of number of producer species. The top model uses resource-based competition models (adapted from Tilman et al., 1997b), while the lower model uses ecosystem-based models employing nutrient depletion at the rooting zone as the basic mechanism for producer competition (adapted from Loreau, 1998). Productivity in Loreau's model was represented by the symbol Φ. In the bottom figure, the dashed lines on either side of the solid line represent variations in the order of species additions (based on intensity of resource use) to the ecosystem. Although one models standing biomass after a given number of generations and the other measures productivity, note the similarity of predictions as well as striking similarity to what has been observed in experimental field studies (Fig. 8). In general, an asymptotic relationship between species richness and production is predicted.

Other models have examined the relationship between species richness and predictability of ecosystem functioning. Hughes and Rougharden (1998) used discrete-time, density-dependent, Lotka-Volterra interspecific competition models to demonstrate that aggregative properties, such as the total biomass in the system (equated with the total density of individuals in the two competing populations), is insensitive to variation in the strengths of interactions (specifically, a_{ij} in traditional Lotka-Volterra models) between species. The model does not directly address the relationship between biomass production (ecosystem functioning) and diversity, but indirectly shows that the "stability" of biomass production is a function of asymmetry of interaction strengths in multi-species assemblages.

In contrast to these models, Doak et al. (1998) showed that aggregative properties of ecosystems, such as biomass production, invariably show lower variance due to statistical averaging, an effect that has come to be informally known as the *portfolio effect* because it is based on the well-known observation that a diverse stock portfolio has more stable long-term yields than portfolios of fewer stocks. That is,

$$CV_{cb} = CV_s k^{-1/2},$$

where CV_{cb} is the coefficient of variance of the total biomass of the community, CV_s is the coefficient of variance of biomass of the individual species, and k is the number of species. In this model, the mean and variance in biomass of each species is assumed equal, but the inverse relationship between the coefficient of variance in whole-system performance is not affected by relaxation of this assumption.

Finally, Naeem (1998) suggested that a different modeling approach is necessary when considering ecosystem functioning and used reliability engineering models to focus on estimating the probability that a system will provide the desired level of performance over a given unit of time. This approach considers the importance of functional groups, redundancy, and local extinction probabilities. The model treats each species as a "component" in an ecosystem, and extinction of that species is equivalent to "failure" of the component. Reliability of the i^{th} species within the j^{th} functional group (R_{ij}) is

$$R_{ij}(t) = e^{-\lambda t},$$

where λ is the probability of extinction over a short time period and t is the time over which reliability is estimated. By convention, at $t = 0$, $R_{ij} = 1$, and reliability of a component declines exponentially over time.

If species are redundant then the probability that the j^{th} functional group (G_j) will make its contribution to the ecosystem over a unit time is

$$G_j(t) = 1 - \prod_{i=1}^{S_j} [1 - R_{ij}(t)].$$

The reliability of the whole system (H), or *ecosystem reliability*, which contains F functional groups is

$$H(t) = \prod_{j=1}^{F} G_j(t).$$

Thus, reliability of ecosystem functioning is a declining function of the number of serially dependent functional groups in an ecosystem but an increasing function of the number of redundant species within functional groups.

All these models, from the Lotka-Volterra to the reliability engineering models, provide simple means by which some of the basic relationships between diversity and ecosystem functioning can be understood. If we consider that ecosystem functions such as production or pollutant degradation, or that ecosystem reliability or *biological insurance* (the insurance that diversity provides for continued ecosystem services) have obvious economic analogies, they may provide the starting point for economic analyses of the importance of biodiversity and ecosystem functioning from an environmental or human welfare perspective.

Summary

These empirical and theoretical studies make at least two points about the potential role of biodiversity, summarized in Table 2. First, greater diversity may yield greater levels of ecosystem functioning due to increased efficiencies of resource use by more diverse species assemblages. Alternatively, an increase in levels of functioning may be due to the sampling effect. Second, greater diversity may reduce the variability of ecosystem functioning, which then yields more reliable or predictable systems. This effect may be due to the portfolio effect (statistical averaging) or the presence of redundant species, which replace species lost due to local extinction (ecosystem reliability or biological insurance).

Table 2. Summary of empirical and theoretical findings on the role of biodiversity in ecosystem functioning. Both levels and dynamics of biogeochemically defined ecosystem functioning may be affected by variation in biodiversity. The effects can be statistical, due to changes in probabilities of certain combinations of species found coexisting in nature as diversity declines. The effects can also be biological or compositional, where changes in community efficiency or dynamics of resource use may be affected by the specific changes in community composition created by biodiversity loss. Each of these terms are described in greater detail in the text.

ECOSYSTEM FUNCTION	STATISTICAL	BIOLOGICAL (COMPOSITIONAL)
LEVELS	· SAMPLING EFFECT	· GREATER EFFICIENCY OF RESOURCE USE
DYNAMICS	· PORTFOLIO EFFECT · BIOLOGICAL INSURANCE · ECOSYSTEM RELIABILITY	· KEYSTONE · STRENGTHS OF INTERACTION · FOOD WEB STRUCTURE · COMMUNITY STRUCTURE

This research has forced ecologists to synthesize community, food web, and ecosystems ecology and consider more carefully the combinatorial (the number of possible permutations or combinations possible for a set of entities) nature of the problem of biodiversity loss. When there are even just a few species (>10), the numbers of possible alternative compositions (or community and food-web structures) due to biodiversity loss is enormous. It is thus not surprising that ecologists initially provided such a rich diversity of hypotheses when asked what the relationship between biodiversity and ecosystem functioning might be (Fig. 3).

It is important to note that currently there are literally only a handful of experiments and only a handful of theories addressing these issues. Considerable debate surrounds interpretation of these studies (André et al., 1994; Givnish, 1994; Garnier et al., 1997; Grime, 1997; Huston, 1997; Tilman, 1997; Tilman et al., 1997c; Wardle et al., 1997a) and a recent appraisal of experimental designs have revealed important limitations of some designs over others (Allison, 1999). The basic findings, however, irrespective of whether they have been pot experiments, greenhouse experiments, experimental field plots, laboratory microcosms, or theory, agree with one another; biodiversity loss can affect the levels and dynamics of ecosystem functioning.

Much more work needs to be done to amass enough knowledge to provide a more complete picture. Scientists must be necessarily cautious. Further observations, experiments, and theoretical treatments will be necessary before the mechanisms and the development of predictive models can be achieved and current debates be resolved. Until then, disagreement about the mechanisms, details, or proper interpretation of studies will abound, but fundamentally few would disagree that biodiversity affects both levels and dynamics of ecosystem processes and the ecosystem services derived from these processes. Thus, conserving biodiversity benefits us not only for the genetic resources we may continue to derive from our biota and the cultural, recreational, and other values of biodiversity we already recognize, but also as integral parts of ecosystems that may affect their functioning and ultimately the services we derive from ecosystem functioning. In the absence of additional information and because biodiversity loss is difficult and sometimes impossible to reverse, the conservative approach would be to accept that biodiversity loss translates into environmental change.

Environmental Economics: An Ecologist's Perspective

Although environmental analyses of natural capital and services include valuations of biogeochemically defined ecosystem services (Costanza et al., 1997; Pimentel et al., 1997), as mentioned above, the value of biodiversity has primarily been examined from the standpoint of its nature as commodities (e.g., genetic resources or sources for tourism). What I have tried to present in the above discussion and review is both the fundamental ecological principles and contemporary research that outline the complex interplay among communities, ecosystems, and the envi-

ronment. This review presents an ecological perspective in which an ecosystem is viewed as a rigorously defined entity whose functioning is understood in terms of fluxes of energy and materials among highly dynamic populations, their local environment (i.e., local physical and chemical conditions), and linkages with other systems through global exchanges of material.

Human modifications of the biota can change the levels and dynamics of ecosystem processes. Human modifications of external factors (e.g., eutrophication, pollution, global warming, increased UVB radiation) can also change the composition of the biota or change the boundary conditions under which an ecosystem functions. Given how remarkably thin the biosphere is (Fig. 1) compared to the hydro-, litho-, and atmosphere, and how critical it is to local and global processes of material and energy flow, extensive human modification of the earth's ecosystems is a likely place to begin analyzing the environmental consequences of biodiversity loss.

Ecologists do recognize that their science is inextricably linked to humans (Lubchenco et al., 1991; Vitousek et al., 1997). The products of their science, however, may appear to be unrelated to social, political, or economic sciences because what ecologists quantify, measure, and model is hard to translate into economic or social terms. The principle difficulty for an ecologist attempting to consider the process of economic assessments of the environmental role of biodiversity on ecosystems is that they (myself included) are most comfortable with terms such as C:N ratios, mass loss, photon flux, microbial C, population growth rates, or trophic structure. Naively, valuation suggests that one could somehow remove the dial face of our instruments (say, an infrared gas analyzer measuring CO_2 flux) and rewrite them to read in dollars. Even more problematic, as mentioned earlier, some ecosystem services are not readily quantifiable in the same way as ecosystem processes. For example, a forest stand is presumed to "purify" air and a wetland is presumed to "purify" water. For an ecologist, "purify" must be transformed into a quantifiable statement like the net degradation of a toxic substance in g m^{-2} s^{-1}.

The ecologist and the economist or social scientist need to find the common ground between what is measured and studied in each discipline. Here are some suggestions from an ecologist's perspective.

1. Human impacts on keystone or singular species are likely to have the greatest impact on ecosystem functioning and should be weighted to reflect this when evaluating the importance of species in a community. Power et al. (1996) suggested that such species' impacts on ecosystem functioning can be weighted by calculating "*community importance values*" or *CI*s. Formally,

$$CI_i = [(t_N - t_D)/t_N](1/p_i)$$

where t_N = the quantitative property (e.g., production or rate of decomposition) of the intact community, t_D = the quantitative property of the community with the i^{th} species removed, and p_i represents the proportional abundance of the i^{th} species before it was removed. Assessing *CI*s, or performing some other evalu-

ation of relevant weights for species contributions to ecosystem functioning should be made part of the research agenda for joint ecological/economic assessment.

2. Loss of redundant species may have smaller effects on levels of ecosystem functioning until some threshold is crossed, where this threshold is the critical minimal species richness (CMSR) necessary for a functional group to perform its role in an ecosystem (Fig. 12). Individual species within a biota closer to the CMSR may be much more valuable for ecosystem functioning, despite their redundancy, than when they occur in species-rich communities. It is important to realize that a CMSR is a heuristic device that indicates the need to determine the number of critical functional groups necessary for providing humans with their desired ecosystem service and to recognize that an ecosystem that is near this point is not very likely to be stable in the face of local extinction. Man-

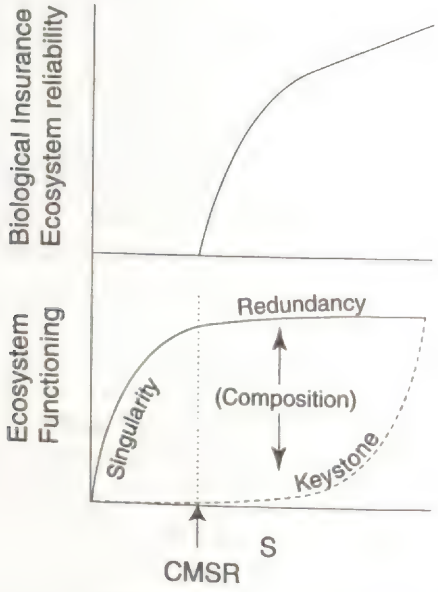


Figure 12. General relationship between biodiversity and ecosystem functioning. Lower plot shows the relationship between ecosystem functioning and biodiversity as currently understood by ecologists. Biodiversity is plotted as the number of species (S). "Singularity" refers to unique or singular species when there is only one species per functional group at low levels of diversity. The critical minimum species richness (CMSR) is where there is at least one species per functional group. Additional species beyond the CMSR are considered redundant. Loss of redundant species does not significantly alter ecosystem functioning; however, some species may be *keystone* species. Loss of keystone species can cause dramatic changes in ecosystem functioning. If diversity consists of both redundant and keystone species, then, depending on the change in community composition, ecosystem response to biodiversity loss can follow any of a variety of trajectories as illustrated by the bi-directional arrow labeled "Composition." The upper plot shows that biological insurance and ecosystem reliability is lowest below the CMSR, but rises quickly even with small amounts of redundancy. (Adapted from Naeem et al., 1998.)

aged ecosystems, for example, are often very close to their CMSR, and it is worth evaluating whether the benefits of working with simple systems truly outweigh the costs of their instability.

3. Redundant species should be considered a source of biological insurance, not just valued as products or commodities. In many cases, it may be impossible to determine definitively that a species does have an insurance value, but in the absence of such information, theory such as the portfolio effect or ecosystem reliability may provide a guideline for determining how to assess the importance of large numbers of redundant species. For example, 11 years of data were necessary before the value of high plant-species richness as a means of drought recovery could be observed in an indirect experimental manipulation of biodiversity (Tilman & Downing, 1994).
4. In the absence of information to the contrary, the prudent strategy is to retain diversity because restoration costs in the face of local extinction may be enormous, or restoration may be impossible if the species is globally extinct. In the same way that economists approach the problem of valuing biodiversity for its potential for new gene products, it is important to recognize biodiversity's potential value for maintaining stable ecosystem functioning.
5. Environmental economics focusing on biodiversity and ecosystem functioning must cross geopolitical boundaries. Humans occupying geopolitically defined regions tend to perceive of their regions as being discrete ecosystems. Ecosystem boundaries seldom match geopolitical boundaries and, as discussed above, are not truly isolated from one another because of materials shared across the lithosphere, hydrosphere, and atmosphere. Thus, though it is possible to imagine island communities economically isolated from the economies of other nations and the globe, this is never true for ecosystem services.

As ecologists and economists continue to make progress on the means and methods for valuing ecosystem services and biodiversity, the linkage between what ecologists and economists do will become more critical. If the Knowledge Revolution becomes an important paradigm for emerging economic and environmental economic theory, procurement, dissemination, and translation of relevant information concerning human welfare, ecosystem services, and environmental health will be dependent on a common knowledge base among the different parties. At some level, the ecologist measuring photosynthesis in the field, the forest manager keeping track of production, the biogeochemist keeping tabs on the global carbon cycle, the climate modelers modeling CO_2 and global warming, and the economists and policy developers working on introducing carbon to global markets are all working on the same problem—the relationship between our biota, the environment, and ourselves. Right now, perhaps the most urgent message to get across to all parties is that the environmental consequences of the widespread changes in our biota need to be better understood by both ecologists and economists.

Acknowledgments

I would like to thank S. Tjossem and an anonymous reviewer for their comments on the manuscript. This effort was supported in part by NSF DEB-9709559.

Literature Cited

Allison, G. W. 1999. The implications of experimental design for biodiversity manipulations. American Naturalist 153: 26–45.

André, M., F. Bréchignac & P. Thibault. 1994. Biodiversity in model ecosystems. Nature 371: 565.

Bulla, L. 1996. Relationships between biotic diversity and primary productivity in savanna grasslands. Pp. 97–120 *in* O. T. Solbrig, E. Medina & J. F. Silva (editors), Biodiversity and Savanna Ecosystem Processes. Springer-Verlag, Berlin.

Butcher, S. S., R. J. Charlson, G. H. Orians & G. V. Wolfe. 1992. Global Biogeochemical Cycles. Academic Press, London.

Carpenter, S. R. 1996. Drivers and dynamics of changes in biodiversity. Pp. 311–318 *in* UNEP (editor), Global Biodiversity Assessment. Cambridge Univ. Press, Cambridge, U.K.

Chapin III, F. S., B. H. Walker, R. J. Hobbs, D. U. Hooper, J. H. Lawton, O. Sala & D. Tilman. 1997. Biotic control over the functioning of ecosystems. Science 277: 500–503.

Chapin III, F. S., O. E. Sala, I. C. Burke, J. P. Grime, D. U. Hooper, W. K. Lauenroth, A. Lombard, H. A. Mooney, A. R. Mosier, S. Naeem, S. W. Pacala, J. Roy, W. L. Steffen & D. Tilman. 1998. Ecosystem consequences of changing biodiversity. Experimental evidence and a research agenda for the future. BioScience 48: 45–52.

Cohen, J. E. 1991. Food webs as a focus for unifying ecological theory. Ecology International Bulletin 19: 1–13.

Collins, S. L. & T. L. Benning. 1996. Spatial and temporal patterns in functional diversity. Pp. 253–282 *in* K. Gaston (editor), Biodiversity: A Biology of Numbers and Differences. Blackwell Science, Oxford.

Costanza, R., R. D'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neil, J. Paruelo, R. G. Raskin, P. Sutton & M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Cronk, Q. C. B. & J. L. Fuller. 1995. Plant Invaders. Chapman and Hall, London, U.K.

Currie, D. J. 1991. Energy and large-scale patterns of animal- and plant-species richness. American Naturalist 137: 27–49.

Daily, G. C. (Editor). 1997. Nature's Services. Island Press, Washington, D.C.

Daily, G. C., S. Alexander, P. R. Ehrlich, L. Goulder, J. Lubchenco, P. A. Matson, H. A. Mooney, S. Postel, S. H. Schneider, D. Tilman & G. M. Woodwell. 1997. Ecosystem Services: Benefits Supplied to Human Societies by Natural Ecosystems. Ecological Society of America, Washington, D.C.

De Angelis, D. L. 1992. Dynamics of Nutrient Cycling and Food Webs. Chapman and Hall, London.

de Groot, R. S. 1992. Functions of Nature. Wolters Noordhoff BV, Groningen, The Netherlands.

Doak, D. F., D. Bigger, E. Harding-Smith, M. A. Marvier, R. O'Malley & D. Thomson. 1998. The statistical inevitability of stability-diversity relationships in community ecology. American Naturalist 151: 264–276.

Drake, J. A., H. A. Mooney, F. di Castri, R. H. Groves, F. J. Kruger, M. Rejmánek & M. Williamson. 1989. Biological Invasions: A Global Perspective. John Wiley & Sons, Chichester, U.K.

Ehrlich, P. R. & A. H. Ehrlich. 1981. Extinction: The Causes and Consequences of the Disappearance of Species. Random House, New York.

Elton, C. S. 1958. The Ecology of Invasions by Animals and Plants. Methuen, London.

Erwin, T. L. 1997. Biodiversity at its utmost: Tropical forest beetles. Pages 27–40 *in* M. L. Reaka-Kudla, D. E. Wilson & E. O. Wilson (editors), Biodiversity II. Island Press, Washington, D.C.

Garnier, E., M.-L. Navas, M. P. Austin, J. M. Lilley & R. M. Gifford. 1997. A problem for biodiversity-productivity studies: How to compare the productivity of multispecific plant mixtures to that of monocultures? Acta Oecologica 18: 657–670.

Gitay, H., J. B. Wilson & W. G. Lee. 1996. Species redundancy: A redundant concept? Journal of Ecology 84: 121–124.

Givnish, T. J. 1994. Does biodiversity beget stability? Nature 371: 113–114.

Grime, J. P. 1997. Biodiversity and ecosystem function: The debate deepens. Science 277: 1260–1261.

Groombridge, B. (Editor). 1992. Global Biodiversity. Chapman and Hall, London.

Hairston, N. G. S., F. E. Smith & L. B. Slobodkin. 1960. Community structure, population control, and competition. American Naturalist 106: 249–257.

Haydon, D. 1994. Pivotal assumptions determining the relationship between stability and complexity: An analytical synthesis of the stability-complexity debate. American Naturalist 144: 14–29.

Hooper, D. U. & P. M. Vitousek. 1997. The effects of plant composition and diversity on ecosystem processes. Science 277: 1302–1305.

Hooper, D. U. & P. M. Vitousek. 1998. Effects of plant composition and diversity on nutrient cycling. Ecological Monographs 68: 121–149.

Horace, R. J. (Editor). 1985. Animal Extinctions: What Everyone Should Know. Smithsonian Institution Press, Washington, D.C.

Hughes, J. B. & J. Roughgarden. 1998. Aggregate community properties and the strength of species interactions. Proceedings of the National Academy of Sciences 95: 6837–6842.

Hughes, J. B., G. C. Daily & P. R. Ehrlich. 1997. Population diversity: Its extent and extinction. Science 278: 689–692.

Huston, M. A. 1997. Hidden treatments in ecological experiments: Re-evaluating the ecosystem function of biodiversity. Oecologia 110: 449–460.

Imhoff, M. L. 1994. Mapping human impacts on the global biosphere. BioScience 44: 598.

Johnson, K. H., K. A. Vogt, H. J. Clark, O. J. Schmitz & D. J. Vogt. 1996. Biodiversity and the productivity and stability of ecosystems. Trends in Ecology and Evolution 11: 372–377.

Jones, C. G., J. H. Lawton & M. Shachak. 1994. Organisms as ecosystem engineers. Oikos 69: 373–386.

Jones, J. A. 1990. Termites, soil fertility and carbon cycling in dry tropical Africa: A hypothesis. Journal of Tropical Ecology 6: 291–305.

Kareiva, P. M., J. G. Kingsolver & R. B. Huey (Editors). 1993. Biotic Interactions and Global Change. Sinauer Associates, Sunderland, Massachusetts.

Klug, M. J. & J. M. Tiedge. 1994. Response of microbial communities to changing environmental conditions and physiological approaches. Pp. 371–378 *in* R. Guerrero & C. Pedros-Alio (editors), Trends in Microbial Ecology. Spanish Society of Microbiology, Barcelona.

Kronberg, B. I. 1993. Response of major North American ecosystems to global change: A biogeochemical perspective. Pp. 121–130 *in* H. A. Mooney, E. R. Fuentes & B. I. Kronberg (editors), Earth System Responses to Global Change. Academic Press, San Diego.

Körner, C. 1993. Scaling from species to vegetation: The usefulness of functional groups. Pp. 117–132 *in* E.-D. Schulze & H. A. Mooney (editors), Biodiversity and Ecosystem Functioning. Springer-Verlag, Berlin.

Kunin, W. E. & J. H. Lawton. 1996. Does biodiversity matter? Evaluating the case for conserving species. Pp. 283–308 *in* K. Gaston (editor), Biodiversity: A Biology of Numbers and Differences. Blackwell Science, Oxford.

Lawler, S. P. & P. J. Morin. 1993. Food web architecture and population dynamics in laboratory microcosms of protists. The American Naturalist 141: 675–686.

Lawton, J. H. 1996. The Ecotron facility at Silwood Park: The value of "big bottle" experiments. Ecology 77: 665–669.

Lawton, J. H. & V. K. Brown. 1993. Redundancy in ecosystems. Pp. 255–270 *in* E. D. Schulze & H. A. Mooney (editors), Biodiversity and Ecosystem Function. Springer-Verlag, New York.

Lawton, J. H. & C. G. Jones. 1993. Linking species and ecosystem perspectives. Trends in Ecology and Evolution 8: 311–313.

Lawton, J. H. & R. H. May (Editors). 1994. Extinction Rates. Oxford Univ. Press, Oxford.

Lawton, J. H., S. Naeem, R. M. Woodfin, V. K. Brown, A. Gange, H. C. J. Godfray, P. A. Heads, S. P. Lawler, D. Magda, C. D. Thomas, L. J. Thompson & S. Young. 1993. The Ecotron: A controlled environmental facility for the investigation of populations and ecosystem processes. Philosophical Transactions of the Royal Society of London, B 341: 181–194.

Leach, M. K. & T. J. Givnish. 1996. Ecological determinant of species loss in remnant prairies. Science 273: 1555–1558.

Levin, S. A. 1970. Community equilibria and stability, and an extension of the competitive exclusion principle. American Naturalist 104: 413–423.

Liebold, M. A. 1996. A graphical model of keystone predators in food webs: Trophic regulation of abundance, incidence, and diversity patterns in communities. American Naturalist 147: 784–812.

Loreau, M. 1995. Consumers as maximizers of matter and energy flow in ecosystems. American Naturalist 145: 22–42.

Loreau, M. 1998. Biodiversity and ecosystem functioning: A mechanistic model. Proceedings of the National Academy of Sciences 95: 5632–5636.

Lovelock, J. 1979. Gaia. Oxford Univ. Press, Oxford.

Lubchenco, J., A. M. Olson, L. B. Brubaker, S. R. Carpenter, M. M. Holland, S. P. Hubbell, S. A. Levin, J. A. MacMahon, P. A. Matson, J. M. Melillo, H. A. Mooney, C. H. Peterson, H. R. Pulliam, L. A. Real, P. J. Regal & P. G. Risser. 1991. The sustainable biosphere initiative: An ecological research agenda. Ecology 72: 371–412.

MacArthur, R. 1955. Fluctuations of animal populations and a measure of community stability. Ecology 36: 533–536.

May, R. M. 1974. Stability and Complexity in Model Ecosystems. Princeton Univ. Press, Princeton.

May, R. M. 1995. Conceptual aspects of the quantification of the extent of biological diversity. Pp. 13–20 *in* D. L. Hawksworth (editor), Biodiversity Measurement and Estimation. Chapman and Hall, London.

McCarthy, T. S. & W. S. Ellery. 1994. The effect of vegetation on soil and ground water chemistry and hydrology of islands in the seasonal swamps of the Okavango Fan, Botswana. Journal of Hydrology 154: 169–173.

McGrady-Steed, J., P. M. Harris & P. J. Morin. 1997. Biodiversity regulates ecosystem predictability. Nature 390: 162–165.

McNaughton, S. J. 1993. Biodiversity and function of grazing ecosystems. Pp. 361–384. *in* E. D. Schulze & H. A. Mooney (editors), Biodiversity and Ecosystem Function. Springer-Verlag, New York.

Menge, B. A., E. L. Berlow, C. A. Blanchette, S. A. Navarrete & S. B. Yamada. 1994. The keystone species concept: Variation in interaction strength in a rocky intertidal habitat. Ecological Monographs 64(3): 249–286.

Mills, L. S., M. E. Soulé & D. F. Doak. 1993. The keystone-species concept in ecology and conservation. BioScience 43: 219–224.

Naeem, S. 1998. Species redundancy and ecosystem reliability. Conservation Biology 12: 39-45.

Naeem, S. & S. Li. 1997. Biodiversity enhances ecosystem reliability. Nature 390: 507–509.

Naeem, S., L. J. Thompson, S. P. Lawler, J. H. Lawton & R. M. Woodfin. 1994. Declining biodiversity can alter the performance of ecosystems. Nature 368: 734–737.

Naeem, S., L. J. Thompson, S. P. Lawler, J. H. Lawton & R. M. Woodfin. 1995a. Biodiversity and ecosystem functioning: Empirical evidence from experimental microcosms. Endeavour 19: 58–63.

Naeem, S., L. J. Thompson, S. P. Lawler, J. H. Lawton & R. M. Woodfin. 1995b. Biodiversity loss in model ecosystems. Reply. Nature 375: 561.

Naeem, S., L. J. Thompson, S. P. Lawler, J. H. Lawton & R. M. Woodfin. 1995c. Empirical evidence that declining species diversity may alter the performance of terrestrial ecosystems. Philosophical Transactions of the Royal Society, London, B 347: 249–262.

Naeem, S., K. Haakenson, L. J. Thompson, J. H. Lawton & M. J. Crawley. 1996. Biodiversity and plant productivity in a model assemblage of plant species. Oikos 76: 259–264.

Naeem, S., Z. Kawabata & M. Loreau. 1998. Transcending boundaries in biodiversity research. Trends in Ecology and Evolution 13: 134–135.

Naiman, R. J., J. M. Melillo & J. E. Hobbie. 1986. Ecosystem alteration of boreal forest streams by beaver (*Castor canadensis*). Ecology 67: 1254–1269.

Orians, G. H. 1997. Biodiversity and terrestrial ecosystem processes. Science Progress 80: 45–63.

Paine, R. T. 1966. Food web complexity and species diversity. American Naturalist 100: 65–75.

Pastor, J., B. Dewey, R. J. Naiman, P. F. McInnes & Y. Cohen. 1993. Moose browsing and soil fertility in the boreal forest of Isle Royale National Park. Ecology 74: 467–480.

Perlman, D. L. & G. Adelson. 1997. Biodiversity: Exploring Values and Priorities in Conservation. Blackwell, Cambridge, Massachusetts.

Perrings, C., K.-G. Mäler, C. Folke, C. S. Holling & B.-O. Jansson (Editors). 1995. Biodiversity Loss: Economic and Ecological Issues. Cambridge Univ. Press, Cambridge.

Peters, R. L., T. E. Lovejoy & M. E. Soulé (Editors). 1992. Global Warming and Biological Diversity. Yale Univ. Press, New Haven.

Pimentel, D., C. Wilson, C. McCullum, R. Huang, P. Dwen, J. Flack, Q. Tran, T. Saltman & B. Cliff. 1997. Economic and environmental benefits of biodiversity. BioScience 47: 747–757.

Pimm, S. L. 1982. Food Webs. Chapman and Hall, London.

Pimm, S. L. & J. H. Lawton. 1977. Number of trophic levels in ecological communities. Nature 268: 329–331.

Pimm, S. L., G. J. Russel, J. L. Gittleman & T. M. Brooks. 1995. The future of biodiversity. Science 269: 347–350.

Polis, G. A. & K. O. Winemiller (Editors). 1996. Food Webs: Integration of Patterns and Dynamics. Chapman and Hall, London.

Power, M. E., D. Tilman, J. A. Estes, B. A. Menge, W. J. Bond, S. Mills, G. Daily, J. C. Castilla, J. Lubchenco & R. T. Paine. 1996. Challenges in the quest for keystones. BioScience 46: 609–620.

Raunkier, C. 1934. The Life Form of Plants and Statistical Plant Geography. Oxford Univ. Press, Oxford, U.K.

Raup, D. M. 1991. Extinction: Bad Genes or Bad Luck? W. W. Norton, New York.

Reaka-Kudla, M. L., D. E. Wilson & E. O. Wilson (Editors). 1997. Biodiversity II. Island Press, Washington, D.C.

Root, R. B. 1967. The niche exploitation pattern of the blue-gray gnatcatcher. Ecological Monographs 37: 317–350.

Rosenzweig, M. L. & Z. Abramsky. 1994. How are diversity and productivity related? Pp. 52–64 *in* R. E. Rickleffs & D. Schluter (editors), Species Diversity in Ecological Communities. Univ. Chicago Press, Chicago.

Sala, O. E., W. K. Lauenroth, S. J. McNaughton, G. Rusch & X. Zhang. 1996. Biodiversity and ecosystem function in grasslands. Pp. 129–150 *in* H. A. Mooney, J. H. Cushman, E. Medina, O. E. Sala & E. D. Schulze (editors), Functional Role of Biodiversity: A Global Perspective. J. Wiley and Sons, New York.

Schlesinger, W. H. 1997. Biogeochemistry, 2nd ed. Academic Press, San Diego.

Schulze, E. D. & H. A. Mooney (Editors). 1993. Biodiversity and Ecosystem Function. Springer-Verlag, New York.

Sisk, T. D., A. E. Launer, K. R. Switky & P. R. Ehrlich. 1994. Identifying extinction threats. BioScience 44: 592–604.

Smil, V. 1997. Cycles of life: Civilization and the biosphere. Scientific American Library, New York.

Soulé, M. E. 1991. Conservation: Tactics for a constant crisis. Science 253: 744–750.

Stork, N. 1997. Measuring global biodiversity and its decline. Pp. 41–68 *in* M. L. Reaka-Kudla, D. E. Wilson & E. O. Wilson (editors), Biodiversity II. Island Press, Washington, D.C.

Tilman, D. 1997. Distinguishing the effects of species diversity and species composition. Oikos 80: 185.

Tilman, D. & J. A. Downing. 1994. Biodiversity and stability in grasslands. Nature 367: 363–365.

Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie & E. Sieman. 1997a. The influence of functional diversity and composition on ecosystem processes. Science 277: 1300–1302.

Tilman, D., C. L. Lehman & K. T. Thomson. 1997b. Plant diversity and ecosystem productivity: Theoretical considerations. Proceedings of the National Academy of Science 94: 1857–1861.

Tilman, D., S. Naeem, J. Knops, P. Reich, E. Siemann, D. Wedin, M. Ritchie & J. Lawton. 1997c. Biodiversity and ecosystem properties. Science 278: 1866–1867.

Tilman, D., D. Wedin & J. Knops. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature 379: 718–720.

Tilman, D., C. L. Lehman & C. Yin. 1997d. Habitat destruction, dispersal, and deterministic extinction in competitive communities. American Naturalist 149: 407–435.

Tilman, D., R. M. May, C. L. Lehman & M. A. Nowak. 1994. Habitat destruction and the extinction debt. Nature 371: 65–66.

Usher, M. B., P. R. Davis, J. R. W. Harris & B. C. Longstaff. 1979. A profusion of species? Approaches towards understanding the dynamics of the populations of the microarthropods in decomposer communities. Pp. 359–384 *in* R. M. Anderson, B. D. Turner & L. R. Taylor (editors), Population Dynamics. Blackwell Scientific, London.

Vitousek, P. M. 1990. Biological invasions and ecosystem processes: Towards an integration of population biology and ecosystem studies. Oikos 57: 7–13.

Vitousek, P. M. & D. U. Hooper. 1993. Biological diversity and terrestrial ecosystem biogeochemistry. Pp. 3–14 *in* E. D. Schulze & H. A. Mooney (editors), Biodiversity and Ecosystem Function. Springer-Verlag, New York.

Vitousek, P. M., H. A. Mooney, J. Lubchenco & J. M. Melillo. 1997. Human domination of Earth's ecosystems. Science 277: 494–499.

Walker, B. 1995. Conserving biological diversity through ecosystem resilience. Conservation Biology 9: 747–752.

Walker, B. 1992. Biological diversity and ecological redundancy. Conservation Biology 6: 18–23.

Wardle, D. A., O. Zackrisson, G. Hornberg & C. Gallet. 1997a. Biodiversity and ecosystem properties. Science 278: 1867–1869.

Wardle, D. A., O. Zackrisson, G. Hornberg & C. Gallet. 1997b. The influence of island area on ecosystem properties. Science 277: 1296–1299.

Whittaker, R. H. 1975. Communities and Ecosystems. Macmillan, New York.

Wilson, E. O. 1988. The current state of biological diversity. Pp. 3–18 *in* E. O. Wilson (editor), Biodiversity. National Academy Press, Washington, D.C.

Zimov, S. A., V. I. Chuprynin, A. P. Oreshko, F. S. Chapin III, J. F. Reynolds & M. C. Chapin. 1995. Steppe-tundra transition: A herbivore-driven biome shift at the end of the Pleistocene. American Naturalist 146: 765–794.

Sustainable Habitation in the Savannas

Brian Walker
CSIRO, Wildlife and Ecology
P.O. Box 84
Lyneham ACT 2602, Australia

Abstract

The savannas have a long history of co-evolution of vegetation, fauna, and people. They are characterized by having strongly seasonal climates and a mixture of grasses and woody vegetation that is strongly influenced by fire and grazing animals. Interpretations of the control of the grass:woody balance have evolved from one of a stable equilibrium combination to an unstable mixture that can be driven for long periods into a state that is so strongly dominated by woody plants that it excludes fire. In the delivery of savanna goods and services most attention has focused on livestock production, as exemplified by the optimal control models described by Perrings and Walker (1997). However, where savannas are used for multiple purposes (food plants, livestock production, livestock for draught, fuel wood, crop production, and perhaps other ecosystem services), a top-down optimization approach is less likely to be successful. It may be better for such savannas to consider them as complex adaptive systems and to try to understand the trajectories the system might follow (in terms of trees, grass, animals, crops, economics, etc.). With an understanding of what influences the directions of possible trajectories it might be possible to learn how to nudge the system away from trajectories that are clearly undesirable, keeping the system changing along those that are more likely to achieve sustainability.

Introduction

In terms of human domination of ecosystems, the savannas of the world rank very high. They have been occupied by people for millennia and, unlike the forests,

people and their livestock live in them, hunt in them, burn them, and use all parts of them, continuously. Apart from a tiny percentage constituting national parks and other protected areas, most of the world's savannas have been sculpted by people. In Africa the process has been one of almost continuous early coevolution—of the vegetation, the fauna (including an uninterrupted presence of megafauna), and the evolving *Homo sapiens*. Most of Africa's national parks have a strong signature of earlier human occupation. In the Americas and in Australia there was coevolution between the vegetation and the fauna, but not with people. Humans only arrived between 20,000 and 50,000 years ago, shortly after which the megafaunas of both continents went extinct, setting in train a new phase of coevolution between the vegetation, the remaining fauna, and people. The "new" savannas so derived were apparently grassier than before, and less nutrient-rich, owing to increased fire frequencies that resulted from the disappearance of the large grazers (Flannery, 1994). In India (the savannas of which share close affinities with those of Africa) there was no sudden Pleistocene extinction of the megafauna, but people and their cattle gradually replaced most of the large herbivores and eliminated most of the large predators.

At this point it is important to note that savannas constitute a very large part of the tropical and subtropical world. They range from almost forest to almost desert. In this chapter I focus mainly on the more arid savannas, the savanna rangelands, used for livestock grazing, but the full array are included.

From the late 1800s the savannas of Africa, America, and Australia were subjected to a sudden and massive disturbance in the form of commercial ranching. Essentially this consisted of introducing artificial watering points to allow for continuous heavy grazing, as opposed to the irregular, seasonal or pulse-grazing which occurs under native herbivores and under the grazing systems used by migratory people and other low-density human populations in Africa. Commercial ranching also introduced nutrient supplementation for livestock, leading to the maximum number of cattle that could be maintained, which in practice meant intensities of grazing dictated only by the welfare of the livestock.

Some concerns were occasionally expressed about the state of the rangeland, but since financial discount rates exceeded perceived rates of any decline in savanna production, little was ever done about it. The result was extensive change in savanna structure and composition over most of the world. The supply of savanna goods and services has changed accordingly, and there is evidence that, indeed, many of them are now less productive (i.e., lower net primary production) than they once were (e.g., Tothill & Mott, 1985). Yet the savannas continue to support millions of people and livestock.

This chapter presents a brief account of what savannas are, followed by their dynamics in response to the dominant drivers. It then considers the supply of goods and services, and concludes with two approaches to managing savannas for sustainability.

Dynamics of Savanna Structure and Function

Tropical savannas have three essential characteristics:

- a mixed vegetation life form of grasses (C4 photosynthetic type) and woody plants (C3), with both playing a significant ecological role;
- a strongly seasonal climate with hot, wet summers and mild, dry winters;
- a periodic fire regime, the frequency increasing from once in 10 or 20 years at the dry end to virtually annual at the wet end.

The composition of any particular savanna depends on its location in the plane of plant available moisture (PAM) and plant available nutrients (PAN) (cf. Frost et al., 1986; Tothill & Mott, 1985). Figure 1 is an example of the distribution of 19 Australian savannas in this plane. Figure 2 shows the relationship between the ratio of woody to grass biomass in these savannas, plotted against a PAM index that incorporates both total PAM and the ratio of the amount of rainfall that gets through to the subsoil to the amount that is retained in the top 25 cm. An increase in this ratio (i.e., more subsoil water, as occurs in sandy soils) tends to favor woody plants, which have proportionally more of their roots deeper in the soil than do grasses (Knoop & Walker, 1985). The distribution of the 19 sites in Figure 2 illustrates two important points: First, "woodiness" in savannas increases both with increasing rainfall and with an increase in the ratio of subsoil to topsoil moisture (as predicted). Second, the evolutionary history of the savanna plays a significant role in actual vs. potential structure, shown by the positions of the pure grassland sites. The general slope of the relationship in Figure 2 predicts that most of these sites

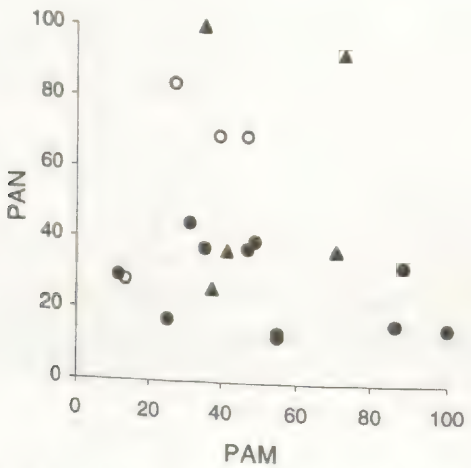


Figure 1. Distribution of 20 Australian savannas in PAM/PAN space where PAM is defined as rainfall/potential evapotranspiration and PAN is the plant available nutrient index for the subsoil taking available phosphorous into account. Natural grasslands are unfilled circles, wooded savannas are black-filled circles, and cleared sites are triangles. (Redrawing from Walker & Langridge, 1997.)

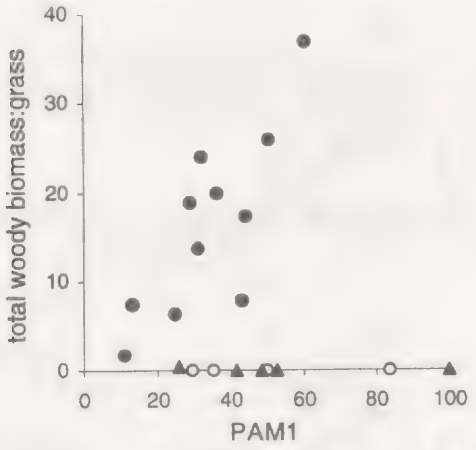


Figure 2. Woody biomass:grass biomass ratio against an integrated PAM index for 20 Australian savannas. PAM1 incorporates both the duration and amount of soil moisture, and the relative difference between the topsoil and the subsoil. Natural grasslands are unfilled circles, wooded savannas are black-filled circles, and cleared sites are triangles. (Redrawn from Walker & Langridge, 1997.)

should have a high woody biomass. In fact, the sites are representative of the vegetation on the cracking clays (the Mitchell grass plains), and it turns out that these pure grasslands are only treeless because no Australian woody species are adapted to growing in cracking clays. Much of the plains are now being invaded by *Acacia nilotica*, a tree introduced in the 1930s from Pakistan to provide shade for livestock; the woody biomass of the invaded sites fits the prediction from Figure 2.

Natural and human drivers of change

Change in savannas is driven by fluctuations in rainfall and by changes in the regimes of fire and grazing. Fire and grazing are natural phenomena in savannas, and their absence constitutes as much of a disturbance as do human-induced increases in their frequencies and intensities. "Natural" regimes of fire and grazing seldom if ever occur any more, and it is extremely difficult to establish what a natural regime would be. Even in national parks the incidence of fires coming in from the outside greatly complicates a policy of allowing "natural" fires. Whatever it may have been it is most unlikely to have been constant or regular. Human-induced changes in fire and grazing, interacting with natural fluctuations in climate, have induced significant changes in most savannas. The most significant of these include the loss of species, particularly perennial grass species, and the development of persistent woody thickets ("bush encroachment" or "woody weed invasion"). The latter dynamics have been the subject of considerable work, and Figure 3 presents a developmental view of the interpretation of the grass-woody balance in savannas. It begins (Fig. 3a) with a (incorrect) simplistic interpretation of a savanna as a single, stable equilibrium balance of the two (Walker & Noy-Meir, 1982) based on the Walter (1978) hypothesis of vertically separated rooting systems. This inter-

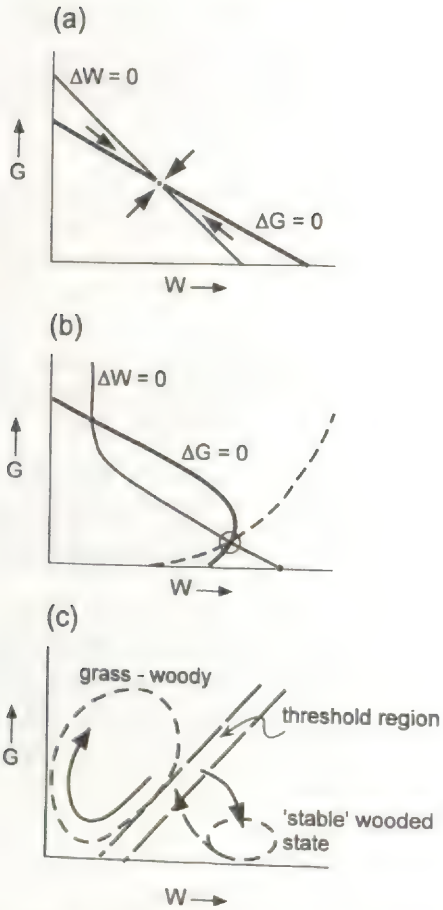


Figure 3. Three progressive models of the dynamics of grass and woody vegetation in a savanna (see text for explanation).

pretation excludes the effects of both grazing and fire. Figure 3b (Walker et al., 1981) extends the model to include grazing effects and allows for alternative stable states. However, the threshold effect required for the two domains of attraction were later (Knoop & Walker, 1985; Scholes & Walker, 1993) shown to apply only under particular, limited soil conditions.

Figure 3c approaches more closely what actually happens (but is probably still incorrect) and includes both grazing and fire. It suggests that under natural conditions grazing is light to moderate and allows for the buildup of grass fuel, especially during good rainfall years, allowing for periodic fires, which maintain a high grass:woody ratio. A well-developed grass layer reduces the amount of water that gets through to the subsoil, where trees have more roots than do grasses, and so reduces the competitive advantage of trees. Furthermore, a vigorous grass layer prevents the establishment of woody seedlings (Knoop & Walker, 1985). Persistent heavy grazing has three effects: (i) it reduces the competitive effect of grass

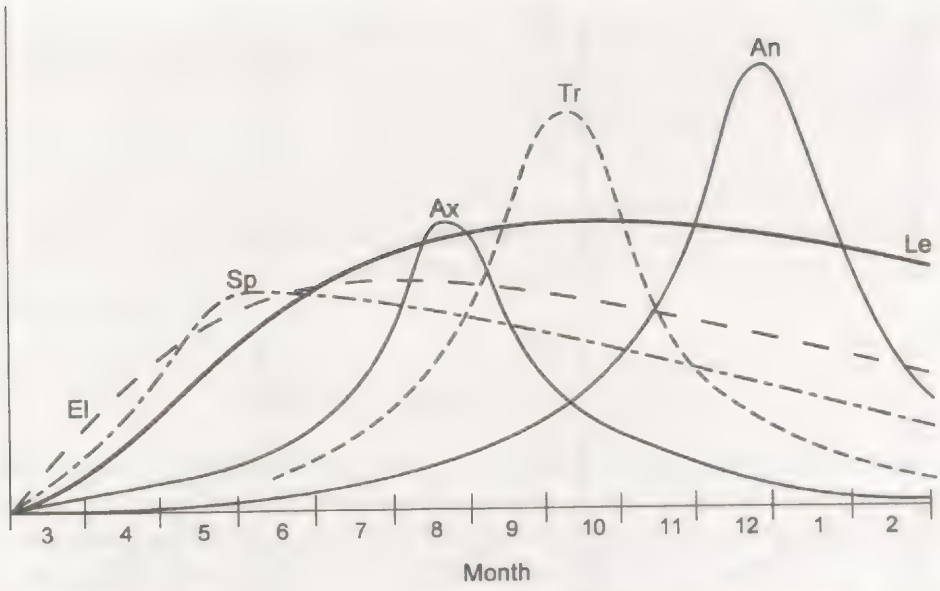


Figure 4. The annual cycle of the green biomass of the six dominant grasses in the seasonal Venezuelan Llanos. El = *Elyonurus adustus*; Sp = *Sporobulus cubensis*; Le = *Leptocoryphium lanatum*; Ax = *Axonopus canescens*; Tr = *Trachypogon vestitus*; An = *Andropogon semiberbis*. (Redrawn from Sarmiento & Monasterio, 1983.)

on woody growth, (ii) it allows the establishment of woody seedlings, and (iii) it prevents the buildup of grass fuel for a fire of sufficient intensity to kill developing (and some developed) woody plants. Once a cohort of woody plants becomes established, it develops into a woody thicket that prevents grass buildup even in the absence of grazing. The thicket state persists until inter-tree competition results in a restructuring and thinning of the woody layer, such that the grass layer can again develop and play a significant ecological role (through competition and fire). This may take a few decades (see the example in Ludwig et al., 1997).

A high diversity of perennial grass species has been shown by McNaughton (1985) to reduce the inter-annual variability in grass production in the Serengeti. A mechanism for this effect, suggested by Silva (1987) for South American savannas, is the loss of phenological diversity. He suggested that precocious palatable grasses are rapidly eliminated under constant heavy grazing. Figure 4, from Sarmiento and Monasterio (1983), shows the growth phenology of the main perennial grasses in the llanos in Venezuela. It illustrates the functional significance of grass species diversity and how loss of species through grazing can influence the seasonal pattern of grass production. Menaut and Cesar (1979) have shown that even in the absence of grazing the early-growing grasses are out-competed by "strong" mid-season perennials, but re-appear following fire. Figure 5 shows the patterns of grass species contributions to biomass from three sites in a Zimbabwe savanna (which has co-evolved continuously with large herbivores) and three sites in an equiva-

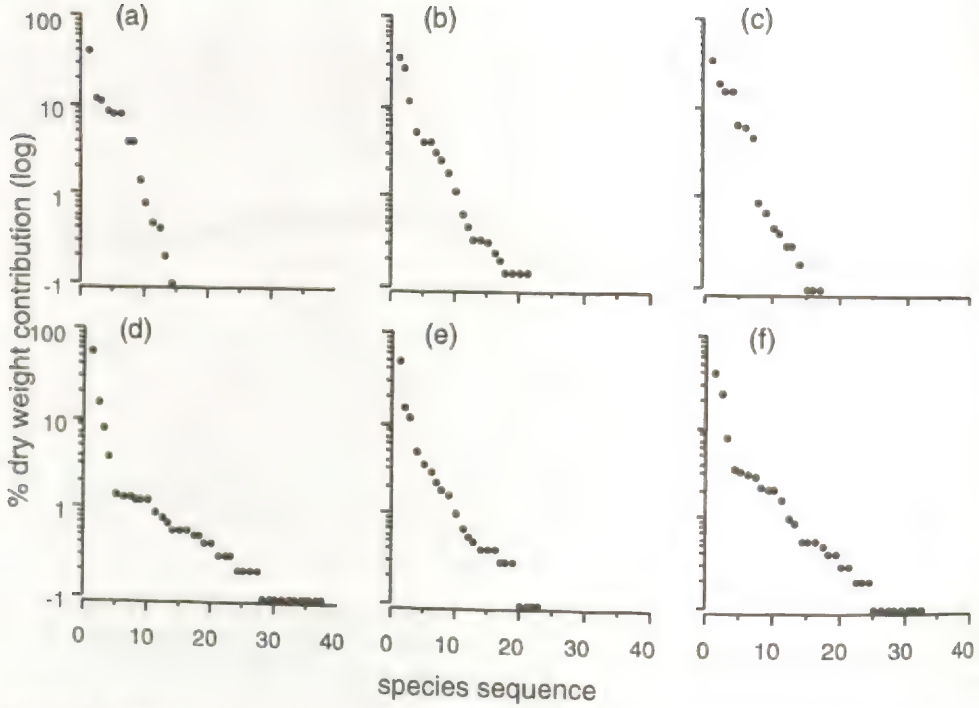


Figure 5. Herbaceous species richness and diversity, as represented by the percentage contribution each species makes to total sward biomass, in decreasing order in three Australian savannas (a–c) and three savanna sites in southeastern Zimbabwe (d = ungrazed, e = lightly grazed by wildlife, f = moderately grazed by cattle). (After Walker et al., 1997.)

lent Australian savanna (from which large herbivores were absent for about 40,000 years, until they were suddenly and heavily grazed around the turn of this century). The Zimbabwe savanna has more species and a greater potential for dampening inter-annual variation in production in response to seasonal rainfall variation (Walker et al., 1997; Walker & Langridge, 1997).

Spatial pattern and dynamics

Spatial patterning in all savannas can arise from the effects of fire and grazing. In the drier savannas, however, it is particularly evident because vegetation cover on the soil is incomplete and results in run-off of rainfall, which leads to a strongly patterned landscape. Tongway and Ludwig (1990) described how the run-off/run-on dynamics result in a source-transfer-sink system reflected in a banded vegetation pattern (see also Ludwig et al., 1997). The effects of this spatial patterning are:

(i) an efficient use of water and nutrient resources,
(ii) a more diverse plant species composition and a more diverse and complex habitat,

(iii) more persistent seasonal growth in the run-on bands than would occur in the absence of this pattern.

Persistent heavy grazing leads to run-off water channeling through the vegetation bands, rather than filtering through them, with a consequent loss of the banded pattern and a net loss of species and production. A decline in the intensity (graininess and strength) of landscape pattern reflects a loss of landscape function processes and this, in turn, reflects a decline in both the productivity and biodiversity status of the savanna.

Savanna Goods and Services

In the context of this symposium the supply of goods and services from savannas reflects the fact that savannas are inhabited by a mix of commercially orientated and subsistence-level people. A full list of all savanna goods and services would be very long, but typically they include the following:

Goods

Fodder for livestock and wildlife—overall quantity, quality, and constancy in inter- and intra-seasonal supply being of concern

Livestock products—meat, milk, dung, blood, hides, and sale of animals

Wildlife products—including, for example, meat and hides from ungulates and reptiles, and invertebrates such as termites, honey ants, Lepidopteran larvae (e.g., "mopane worms" in Africa and "witchety grubs" in Australia) and locusts

Edible plants and fruits

Medicinal plants

Housing material—poles and thatch, and dung for floors

Fuel—wood and dung

Services

Regulated hydrology (permanent streams and fewer large floods)

Maintenance of soil fertility

Tourism

Carbon storage

Draught (use of livestock for ploughing)

Pest control (source of invertebrate predators for crop insect pests)

With the exception of those areas where woody plant invasion is regarded as a rangeland problem, and therefore where carbon storage is increasing, the present trend in human impact on all the above goods and services is negative, due to both an overall increase in intensity of use and to inappropriate management practices. Managing to maximize short-term returns from one particular product (usually live-

stock sales) has negative effects on the other goods and services, and can also reduce the long-term sustainability of livestock production itself. Overall intensity of use is related to increasing human numbers—the trend in the savannas of Africa and Latin America.

The economic viability of commercial livestock production in the world's rangelands has been declining in recent decades, and in most regions pure livestock production is at best a marginal proposition (Walker, 1995). For most people who live in the rangelands the goal is no longer (if it ever was) maximum production, or even maximum sustainable production. It has become sustainable habitation (Walker, 1996; and see Abel & Ryan, 1998), and sustainable habitation in savanna rangelands depends on some combination of multiple use. How to approach this in a scientific way, including the necessary ecological, economic, and social requirements for sustainability, is a major challenge for managing the savannas.

Managing Savannas for Sustainable Habitation

As envisaged in this chapter sustainable habitation includes both the persistence of the ecological state of the savanna (including inter-annual variations, but with the property of returning to the same state given the same environmental conditions) and the persistence of the human "enterprise" (i.e., economic and social sustainability). Managing savannas to achieve sustainable habitation, defined in this way, can be approached in a top-down or a bottom-up way. The former is the more conventional and includes all the land-use planning and optimization approaches. An example follows:

Perrings and Walker (1997) presented an optimal control approach to the management of a savanna rangeland in which the goal of management is to maximize some index of welfare from the use of livestock, grass, and shrubs through choice of the level of offtake of livestock. The level of offtake is determined by maximizing the net present value of a function that includes the market price of livestock and the costs of maintaining the herd, modified by an appropriate discount factor, all subject to the equations for the dynamics of livestock, grass, and shrubs. These equations include the competitive effects of grass and shrubs on each other, and the effects of fire in controlling shrubs. If too much grass is grazed there is insufficient buildup of fuel for a fire, and the system develops into a shrub-dominated state with very low livestock production. The optimization process involves calculation of the optimal grazing pressure at any one time and then adjusting the grazing pressure to this level through the level of offtake of livestock. The optimal grazing pressure (for all but very high discount rates) ensures that there is sufficient grass fuel to control the increase in shrubs and thereby maintain the productivity of the grass layer.

Figure 6a–c presents three sequences of grass, shrub, and livestock biomass corresponding to an uncontrolled grazing system (a), an optimally controlled system allowing for fire to maintain grass (b), and the same system in which it is as-

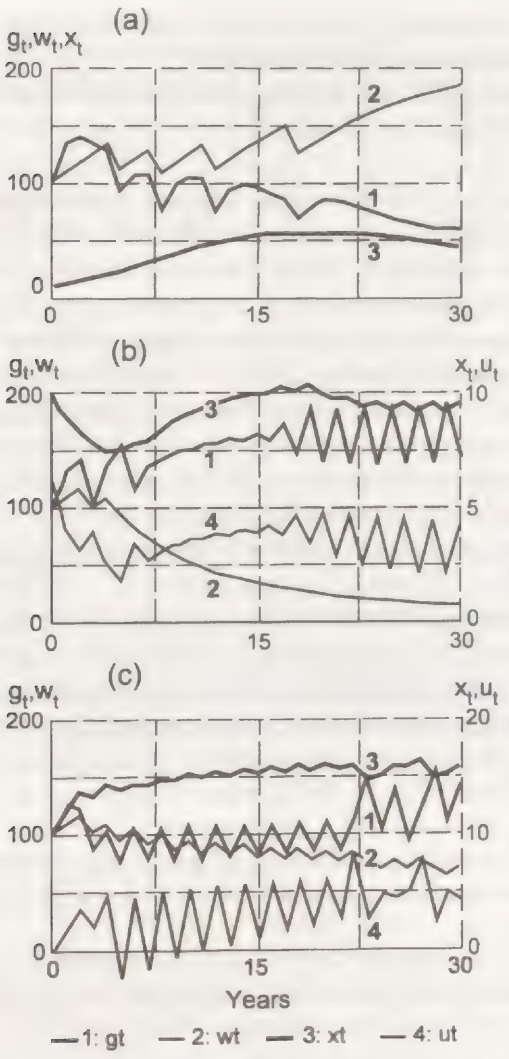


Figure 6. Dynamics of grass (g), wood (w), livestock (x), and livestock offtake (u) in an uncontrolled savanna (a), a savanna under an optimal offtake policy allowing for fire (b), and the same system as in (b) but assuming that conversion of grass into livestock biomass is less efficient (c) (i.e., higher grazing pressure per unit of livestock biomass) (after Perrings & Walker, 1997).

sumed that grazing pressure per unit of livestock biomass is much higher (i.e., conversion of grass into livestock biomass is less efficient) (c). The three patterns are quite different. Figure 6a is all too commonly observed in the savannas and reflects the lack of appreciation of the grass-woody dynamics on the part of many managers. Their recognition that shrub densities are too high comes too late to allow for control by fire. If managers include the effects of these dynamics in an optimized way, then long-term (sustainable) use of the savanna for livestock production is achievable (Figure 6b). However, the difference between Figure 6b and

6c emphasizes the need to get good estimates of the parameters in the equations if this approach is to be successfully applied.

This last point (the significant difference in optimal management due to parameter estimates) highlights a major problem with the top-down optimal control approach. Given the consequences of getting the estimates wrong, the likelihood of successful outcomes, even in developed countries, is not high. It is a useful and potentially valuable approach for a single enterprise where management is good and where there is an opportunity to do the preliminary measurement and modeling work. But in many circumstances a less prescriptive, bottom-up approach is likely to be more successful, especially where there are a number of different forms of land use. In subsistence farming areas, for example, livestock production, fuel wood use, and crop production interact, both complementing and competing with each other. Under such conditions a top-down optimization model attempting to maximize some complex welfare function is unlikely to offer the best approach to finding the way toward sustainable habitation.

The multiple land use system involving subsistence agriculture (arable crops), trees, grass, and livestock, varying in their proportions and in their spatial distribution over time, constitutes a very complex system. From the perspective of how best to approach analyzing and managing such a system a key question is: Do the components of the system conform to the criteria for a complex adaptive system, as suggested by Levin (1998)? Levin's three criteria are:

- diversity and individuality of components
- localized interaction among the components
- an autonomous process that selects from among the components (based on the results of local interactions) a subset for replication or enhancement

The subsistence savanna system operates at two spatial scales. The higher level, coarse scale is a mosaic made up of two basic types of components, each of which can be in a number of states. The two basic components are (i) the "natural" savanna (consisting of grass, shrubs and trees, and livestock) and (ii) arable crop lands. The management actions at the fine scale include fire, livestock harvest, firewood harvest, and various tillage and cropping practices. The net effect is savanna patches in various states, and various kinds and conditions of crops. At the mosaic scale the management decisions involve clearing land for crops and allowing cropped land to go fallow, based on the perceived relative values of the crops, livestock, and other savanna goods and services. The net effect is a changing pattern of savanna and crop patches.

It seems plausible that the components of this multi-scale system could well meet the first two of Levin's criteria. The autonomous selection process, however, is more difficult to identify. But if it exists, it is unlikely to be a single factor. It would more likely be a combination involving climate conditions and the relative magnitudes of market prices for livestock, market prices of crops, and the value to the managers of the other savanna goods and services. The combination of these three

values, interacting with the current climatic conditions, acts to "select" the proportions of the savanna under natural vegetation and under crop land, at the broad scale, and the stocking rate of livestock, use of fire, choice of crop type, tillage practices (partly a function of available livestock for draught), etc., at the patch scale.

If we accept the proposition that the spatial dynamics of arable crops and savanna patches conform to a complex adaptive system, then a top-down optimal control model is unlikely to achieve an enhanced probability of sustainable habitation. A bottom-up approach, involving an increasing understanding of the interactive dynamics of the biophysical and socioeconomic processes, in which managers learn how the system responds to various management interventions in combination with various combinations of environmental conditions, has a better chance of success. Such a developing understanding allows managers to nudge the system along trajectories that indicate sustainability, rather than trying to force it into some particular pattern. The question is: *Do* human occupied savannas (i.e., soils, trees, grass, livestock, crops, people, and economics) conform to the behavior of complex adaptive systems?

Personal observation of the dynamics of savanna lands under subsistence agriculture suggests that they show both elements of controlled, planning behavior, and elements of the behavior of complex adaptive systems. It is therefore a moot point whether their chances of achieving sustainable habitation would be enhanced more through a top-down planning process based on some form of optimization, or through the bottom-up piggyback approach. It is certainly worth exploring the options further.

Literature Cited

Abel, N. & S. Ryan (Editors). 1998. Sustainable Habitation in the Rangelands: Proceedings of a Fenner Conference on the Environment, Canberra, October 1996. CSIRO, Australia.

Flannery, T. 1994. The Future Eaters. Reed Books, Sydney, Australia.

Frost, P. G. H., E. Medina, J.-C. Menaut, O. Solbrig, M. Swift & B. Walker. 1986. Response of savannas to stress and disturbance. A proposal for a collaborative programme of research. IUBS, Special Issue 10, Paris, France.

Knoop, W. T. & B. H. Walker. 1985. Interactions of woody and herbaceous vegetation in a southern African savanna. Journal of Ecology 73: 235–253.

Levin, S. A. 1998. Ecosystems and the biosphere as complex adaptive systems. Ecosystems 1: 431–436.

Ludwig, D., B. Walker & C. S. Holling. 1997. Sustainability, stability, and resilience. Conservation Ecology [online] 1(1): 7. Available from the Internet. URL: http://www.consecol.org/vol1/iss1/art7

Ludwig, J., T. Tongway, D. Freudenberger, J. Noble & K. Hodgkinson (Editors). 1997. Landscape Ecology, Function and Management: Principles from Australian Rangelands. CSIRO Publishing, Collingwood, Victoria.

McNaughton, S. J. 1985. Ecology of a grazing ecosystem: The Serengeti. Ecological Monographs 55(3): 259–294.

Menaut, J.-C. & J. Cesar. 1979. Structure and primary productivity of Lamto savannas, Ivory Coast. Ecology 60: 197–1210.

Perrings, C. & B. Walker. 1997. Biodiversity, resilience and the control of ecological-economic systems: The case of fire-driven rangelands. Ecological Economics 22: 73–83.

Sarmiento, G. & M. Monasterio. 1983. Life forms and phenology. Pp. 79–108 *in* F. Bourliere (editor), Tropical Savannas. Elsevier, Amsterdam.

Scholes, R. J. & B. H. Walker. 1993. An African Savanna. Cambridge Univ. Press, Cambridge, U.K.

Silva, J. F. 1987. Responses of savannas to stress and disturbance: Species dynamics. Pp. 141–156 *in* B. H. Walker (editor), Determinants of Tropical Savannas. IRL Press, Oxford, U.K.

Tongway, D. J. & J. A. Ludwig. 1990. Vegetation and soil patterning in semi-arid mulga lands of eastern Australia. Australian Journal of Ecology 15: 23–34.

Tothill, J. C. & J. J. Mott. 1985. Ecology and Management of the World's Savannas. Australian Academy of Science, Canberra and C.A.B., Farnham Royal, Bucks, U.K.

Walker, B. H. 1995. Having or eating the rangeland cake. Proceedings of the Fifth International Rangeland Congress, Salt Lake City, Utah, July 1995. 22–28.

Walker, B. H. 1996. The concept of sustainable habitation. Pp. 11–13 *in* N. Abel & S. Ryan (editors), Sustainable Habitation in the Rangelands: Proceedings of a Fenner Conference on the Environment. Canberra 29th–30th October 1996. CSIRO, Gungahlin.

Walker, B. H. & J. L. Langridge. 1997. Predicting savanna vegetation structure on the basis of plant available moisture (PAM) and plant available nutrients (PAN): A case study from Australia. Journal of Biogeography 24: 813–825.

Walker, B. H. & I. Noy-Meir. 1982. Aspects of the stability and resilience of savanna ecosystems. Pp. 556–590 *in* B. J. Huntley & B. H. Walker (editors), Ecology of Tropical Savannas. Springer-Verlag, Berlin.

Walker, B. H., D. Ludwig, C. S. Holling & R. M. Peterman. 1981. Stability of semi-arid savanna grazing systems. Journal of Ecology 69: 473–498.

Walker, B. H., F. R. McFarlane & J. L. Langridge. 1997. Grass growth in response to time of rainfall and season along a climate gradient in Australian rangelands. The Rangeland Journal 19(1): 95–108.

Walter, H. 1978. Ecology of Tropical and Subtropical Vegetation. J. H. Burnett (editor). [Translated by D. Mueller-Dombois.] Oliver and Boyd, Edinburgh.

Forest Ecosystems Functioning, Threats and Value: Mexico as a Case Study

Rodolfo Dirzo
Northern Arizona University,
Biological Sciences, P.O. Box 5640
Flagstaff, Arizona 86011, U.S.A.
Current address: Instituto de Ecología, UNAM,
A.P. 70-275, Mexico 04510 DF

Abstract

The relationships between biodiversity, human dominance of ecosystems, and the ecological consequences of human impact on ecosystem services are illustrated with the case of forest ecosystems of Mexico—a megadiversity country. A significant proportion of Mexico's megadiversity is accounted for by the biodiversity of forests. Such biodiversity implies a wealth of biological resources and ecosystem services. Yet, despite their biological richness, Mexican forests are significantly threatened, particularly by current land-use patterns; climatic changes seem to threaten mostly the temperate but not tropical forests. The magnitude of such threats is documented with estimates of plant species loss due to deforestation, carbon emissions due to land use, and the loss of ecological interactions due to defaunation. It is argued that current threats stem from a lack of appreciation of the multiple functions, services, and of the total economic value of forests. A recent exercise indicates that the total economic value of Mexican forests is very high, due largely to their functional value (carbon sequestration and watershed protection). In order to capture the full value of forests, sustainable management practices are necessary; these are discussed, paying emphasis on those located toward the center of a seemingly antagonistic gradient (tension) between biodiversity conservation and commercial exploitation.

Introduction

Forest ecosystems constitute the major terrestrial biome, encompassing a latitudinal span from the tropics to close to the poles, and elevational ranges from sea level to over several thousand meters. Collectively, they occupy an area of about 354 billion hectares, or about a third of the land surface (Noble & Dirzo, 1997). Throughout this vast eco-geographic expanse, forests change in their physiognomy (and functional properties), depending on latitude and elevation, from the tropical forests, to the deciduous forests, to the conifer forests and mixed forests in the areas of contact of these broad categories. Forests contain a large fraction of the Earth's biological richness, including, for example, close to 70% of the planetary plant species (given that tropical forests alone harbor some 180,000 species: Raven, 1992), and account for the vast majority of the terrestrial carbon stored in terrestrial biomass pools. Tropical evergreen forests alone, which comprise only 11% of the Earth's land area, have accounted for ca. 60% of the carbon in aboveground biomass pools of the world's forests and 37% of the terrestrial net primary productivity of the Earth (Whittaker & Likens, 1975; Potter et al., 1993; Dixon et al., 1994). Collectively, forest ecosystems have the potential to sequester 1 to 3 Gt of carbon for about a century (Dixon et al., 1994). Clearly, forests are a critical component of the Earth's life-supporting systems.

In order to illustrate in more detail the biological importance, functioning, threats, and value to society of forest ecosystems, I will describe the forests of Mexico. My justification to address the case of Mexico is that it is recognized as one of the megadiversity regions of the planet, and contains most of the forest types of the world. In addition, they face formidable threats, similar to those of other developing countries, where biological diversity is concentrated (see Wilson, 1992). In this paper I will first provide a summarized description of the biological characteristics of Mexican forests. I will then discuss some of the actual and potential threats; in particular I will try to illustrate the effects of climatic change and land use and how these in turn threaten some functional properties of forests. Finally, I will discuss some components of the economic value of Mexico's forests, and some policy considerations for sustainable management practices.

Biological Characterization

One aspect that defines the megadiversity status of Mexico is its species richness, containing approximately 10% of the global total known species of several groups of plants and animals (see Ramamoorthy et al., 1993). Such constellation of species is not distributed homogeneously; rather, it is accommodated ecologically, according to forest types. Major vegetation types in Mexico can be grouped, for the sake of simplicity, into five types: arid and semi-arid ecosystems, temperate coniferous and broadleaf forests, tropical cloud forests, and tropical dry and ever-

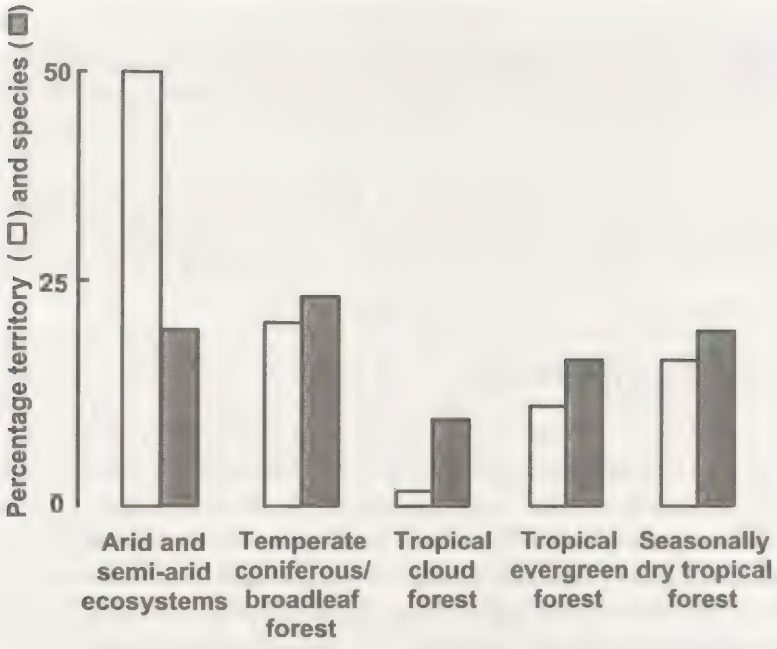


Figure 1. Percentage of the national territory (□) and the number of plant species (■), corresponding to the flora of Mexico, for the five major ecosystems present in the country. The total territory is 2,000,000 km^2, and the total number of species for the Mexican flora is the estimated number of 30,000 (Toledo, 1988). (Derived from Rzedowski, 1993.)

green forests (Rzedowski, 1993). The proportion of the Mexican territory covered by each of these vegetation types shows a gradient as follows (Fig. 1): arid/semiarid ecosystems>temperate forests>tropical dry forests>tropical evergreen forests>tropical cloud forests. On the basis of a species-area relationship argument, a similar gradient of plant species diversity should be expected. Using the published information on several regional floras and his own personal experience, Rzedowski (1993) estimated the relative contribution of each of the major ecosystems to the total species richness of the country. On the basis of this and using Toledo's (1988) widely cited figure of 30,000 species for the Mexican flora, a pattern of species richness distribution as a function of ecosystem type can be obtained (Fig. 1). In contrast to the territory, floristic diversity shows a gradient whereby species richness is concentrated on the forests. In addition, tropical vegetation has the greatest over-representation of species, particularly the cloud forests and the tropical rainforests. In other words, the most diverse biomes are forests, particularly the tropical forests, and the least diverse are the non-forest biomes. Next I describe the forests in more detail.

Conifer forests

These forests contain an impressive diversity for the standards of conifer forests worldwide (Dirzo, 1994). For example: of the 100 species of pines in the world,

Mexican forests contain 50. These include pines adapted to living at sea level; pines adapted to living at the elevational limit of arboreal vegetation, at ca. 4000 m; pines in very wet localities; and pines in arid zones such as the Mexican pinyon pines. A total of 26 pine species are endemic to Mexican forests, some of them of very restricted distribution. In addition to pines, other conifers are also present, as well as a rich understory flora (Rzedowski, 1993; Dirzo, 1994). Besides the valuable resources these forests contain in the form of species and gene pools, their functional role is quite obvious, although poorly quantified, particularly in terms of control of erosion and flooding, and carbon storage.

Temperate broadleaf forests

These forests, largely located in mid elevations, have as a highlight an incredible diversity of oak tree species, including a total known of 180 species, equivalent to 45% of the total species of the world (Dirzo, 1994). As in the case of the pines, many oak species have a distribution restricted to Mexico, and include species remarkably adapted to a range of ecological scenarios, from sea level, in close association to tropical forests, to species typical of humid and high-elevation forests. Ecosystem-function roles of these temperate forests are similar to those of conifer forests.

Quite frequently pines and oaks occur together, forming the pine-oak mixed forests of mid elevations. In conjunction, 70% of the species present in pine-oak forests in Mexico are endemic.

Cloud (montane) tropical forests

These forests are largely unexplored in general, and in Mexico in particular. They are characterized by their incredible mixture of species, including, for example, oaks *(Quercus* spp.), sweet gum trees (*Liquidambar* spp.), and other typically temperate taxa (*Juglans* sp., *Ostrya* sp., etc.), in direct association with trees such as *Cecropia* sp., many lauraceous, and other very typical tropical trees. Even though these forests occupy ca. 1.5% of the Mexican territory, their contribution to the floristic diversity of the country is close to 10%, including a disproportionately large number of endemic species (Rzedowski, 1993). A salient aspect of these forests is their crucial role in the catchment of humidity and as reservoirs and provisioning of water. Given their ecological affinity, it is not surprising that these cloud forests contain the largest densities and diversities of species of ferns and, particularly, of epiphytes (Dirzo, 1994), many of them largely unknown as to their formal identity, let alone their functional and applied-type properties.

Tropical evergreen (rain) forests

Several aspects are notable to these forests in Mexico. Some are quite exclusive to Mexico and some are rather universal. The universal aspects include their great biological diversity, despite the fact that in Mexico they reach their northernmost distribution on the continent. In addition, like other tropical rain forests, in Mexico

they exhibit a significant trend of increase in diversity with the decrease in latitude (Dirzo, 1992). In a series of quantitative studies, at different localities in their natural distribution, using the same sampling method of Gentry (1982) to make data strictly comparable, it has been found that floristic diversity increases significantly from north to south. Thus, sites like Chajul, in Chiapas, have approximately twice the number of plant species per unit area than La Hausteca, in San Luis Potosí. Intermediate localities like Los Tuxtlas, in Veracruz, have an intermediate diversity (Dirzo, 1992). This is also reflected at the level of complete floras. For example, the region of Los Tuxtlas contains nearly 1000 species, approximately the same diversity as the whole of the British Isles (even though the latter is several hundred times larger than Los Tuxtlas). Further south, but still within Mexico, tropical rain forest floras are even richer. Lacandonia, for example, contains an estimated total of 4314 species (Martínez et al., 1994).

Ecosystem services provided by these forests include a very large list, from the provision of countless numbers of species of direct value to humans, particularly the non-timber forest products; to control of erosion and prevention of flooding; to regulation of climate and sequestration of carbon (discussed below) and other elements. For example, total aboveground biomass in some Mexican tropical forests is around 400 Mg/ha, containing element pools of around 200 Mg/ha of carbon, and 1700 kg/ha of nitrogen (Hughes, 1997).

Tropical dry forests

Unlike the wetter tropical forest types, which reach their greatest species richness at or near equatorial latitudes, Neotropical dry forests seem to be richest in plant species at subtropical latitudes (Gentry, 1995). Although these forests are much less studied than rain forests in general and in Mexico in particular, the compilation of data by Gentry (1995) insinuated the importance of tropical dry forests at these latitudes for global plant diversity conservation. A recent study by Trejo (1998), in which floristic diversity was sampled in 20 sites throughout the distribution of this biome in Mexico, again using the method of Gentry (1982), highlights the importance of these Mexican forests from the point of view of their diversity. Sampling forests with a standardized protocol throughout the world, Gentry (1982) found that, up to a certain limit, total annual rainfall (TAR) correlates surprisingly well with plant species diversity (S) in a simple linear fashion: the greater the local precipitation, the greater diversity per standardized unit area (0.1 ha). Thus alpha species richness could be predicted from rainfall. Using Gentry's (1988) equation ($S = 0.488(TAR) + 10.2$; $r = 0.834$; $P < 0.01$), the expected number of plant species in each of the studied localities was estimated according to their precipitation. The results showed that, with few exceptions, Mexican dry forest sites have a larger number of species than would be expected according to the general pattern of tropical floristic diversity discovered by Gentry. Moreover, this study revealed that not only were the individual sites highly diverse, but the turnover of species from site to site was also very large. For example, on average, Sorensen's index of species

similarity among all 20 sites was only 9% (range 0–44%). Thus both alpha and beta diversity are very high in Mexican tropical dry forests. Finally, the level of floristic endemism in these forests is extremely high: about 40% of the species occurring in Mexico are endemic to tropical dry forests, a value 8 times greater than the corresponding figure for tropical evergreen forests (Rzedowski, 1993).

From the standpoint of ecosystem functioning these forests are quite poorly studied. Carbon sequestration (discussed below) is an obvious role. Also, research on erosion control by these forests in western Mexico (Maass et al., 1988) indicates that soil loss increases up to 100 times when dry forests are converted to low productivity maize plots. Average soil loss values were 0.2 and 130 tons/ha/year respectively. In concordance with the great diversity of these forests, ethnobotanical evidence highlights their great potential of provision of resources. Studies in five dry forest areas showed that the numbers of plant species used by local inhabitants, mostly for medicinal uses, ranged from 121 to 305 (Bye, 1995).

The Threats to Biological Diversity

Despite their beauty, biological relevance and tremendous wealth as a library of genetic information, much of which is of direct importance to human societies, forests are being drastically affected due to anthropogenic activities. I would like to characterize such threats, on the one hand, in terms of the current patterns of land use, and in terms of the potential changes in extension or coverage to be expected due to climatic change, on the other.

Changes in forest land area

I will use data on deforestation rates and patterns to illustrate the threats imposed by land-use patterns. Deforestation has a long history in Mexico (summarized in Masera et al., 1997), with major pulses soon after the arrival of the Spanish conquistadors; this was followed by significant episodes in the early 1910s due to concessions to enterprises for exploitation of temperate forests, and a major pulse in the 1940s to 1970s, when tropical forests suffered the most, due to development projects and subsidies for cattle ranching development. This history of deforestation led to the contraction of Mexican forests such that, for example, tropical evergreen forests were reduced to about 10% of their original coverage in the late 1970s (Rzedowski, 1978), while tropical dry forests were reduced to 27% of their original extension by the 1990s (Trejo, 1998).

Regarding the most recent situation, according to a study on the patterns of deforestation of Mexican forests in the mid 1980s (Masera et al., 1997) (Table 1), approximately 668,000 ha were being lost per year during the mid 1980s, leading to an overall rate of deforestation of 1.3%/year. Deforestation rates were considerably higher for tropical forests: 1.9%/year for dry forests and 2.0%/year for tropical rain forests, versus 0.64%/year for both types of temperate forests.

Table 1. Estimates of absolute deforestation and deforestation rates by forest type in Mexico's forests for the 1980s. (From Masera et al., 1997.)

	Temperate coniferous	Temperate Broadleaf	Tropical Evergreen	Tropical Deciduous	Total
Deforestation (ha H 10^3/year)	110	57	195	306	668
Deforestation rate (%/year)	0.64	0.64	2.0	1.9	1.29

Table 2. Estimates of deforestation and forest fire shares (percentages of territory) by conversion activity in Mexico's closed forests for the 1980s. (Data for first row in ha × 10^3/year.) From Masera et al. (1997).

Activity	Temperate coniferous	Temperate Broadleaf	Tropical Evergreen	Tropical deciduous	Total
Deforestation and fires	163	82	237	322	804
Cattle grasslands	28%	28%	58%	57%	49%
Agricultural lands	16%	17%	10%	14%	13%
Harvesting of forest products	5%	5%	2%	5%	4%
Forest fires	49%	47%	22%	7%	24%
Other	3%	3%	7%	16%	10%

Adding the area affected by forest fires, it was estimated that a total of 804,000 ha underwent major perturbations in the mid 1980s. Thus considering forest fires and other land uses, the authors of this study calculated the proximal, leading causes of forest conversion (Table 2). Conversion to pastures, to accommodate the continuing expansion of cattle ranching, is the predominant deforestation factor in tropical forests, with values close to 60%. Forest fires and agricultural activities are the predominant factor in temperate forests with values approaching 65%. Conversion to pastures is not negligible in temperate forests either: this land use contributes, overall, 28% of the total conversion of these forests.

A representative example of the consequences of land-use patterns at a regional scale is that of the tropical region of Los Tuxtlas in Veracruz, Mexico (Dirzo & García, 1992). In this region, not only are deforestation rates very high, but the remaining areas of forest appear currently as an archipelago of fragmented patches of forest immersed in a sea of cattle grasslands.

Potential consequences of species loss

A direct expected consequence of the currently predominant patterns of land use is the potential loss of species.

The species-area relationship of the Theory of Island Biogeography (McArthur & Wilson, 1967), in which species number (S) is predicted to increase with the area of habitat (A), depending on the slope (z) of the relationship, can be used to predict the number of species expected to be lost with the reduction in habitat area. However, a limitation to the application of this approach to estimate species loss is

the basic fact that the numbers of species are not known for many sites in which deforestation rates have been estimated with some accuracy.

A recent, thorough survey of the floristic diversity of the extensive Lacandon forest in Chiapas (Martínez et al., 1994) coincides with a detailed analysis of the deforestation of the area (Mendoza & Dirzo, 1999) and permits the estimation of potential plant species extinction. From the deforestation study a simple exponential decay model was applied to predict the remaining vegetation at different times, on the basis of the average current deforestation rate. The expected reduction in species richness based on the species-area relationship was calculated using two values of the slope of the species-area relationship. It was found that, in the worse-case scenario, with a slope of 0.35, by year 2034 when the forest would be expected to be reduced to 50% of its present coverage, up to 22% of the estimated total 4314 species of this rich flora (i.e., 933 species) would be committed to extinction. By year 2135, at current average deforestation rates, the area of Lacandonia would be reduced to 10% and the number and percentage of plant species committed to extinction would be 2389 and 55%, respectively.

While the use of this type of model to predict species extinction remains a controversial issue among conservation biologists, there is very little doubt that a major pulse of species *population* extinction is taking place due to forest reduction and fragmentation. However, we simply do not have enough data to document this major threat to the genetic library of forests, and the resources they provide to human societies (in Mexico or globally).

Forest coverage under scenarios of climatic change

In order to explore the potential response of Mexican forests to climatic change, Villers and Trejo (1997) carried out a study to assess potential changes in distribution arising from climatic change. They first related vegetation patterns to current climate conditions and subsequently generated scenarios of expected climate distribution. The prospective scenarios were constructed with two models based on a doubled atmospheric CO_2 concentration: the Geophysical Fluid Dynamics Laboratory Model (GFDL) and the Canadian Climate Center Model (CCCM). In addition, a scenario was constructed under the assumption of a homogeneous temperature increase of 2°C and a homogeneous decrease of 10% precipitation. The current and future climate-vegetation maps were overlaid to identify the regions of the country where a change is likely to occur under each scenario. The results are summarized in Table 3. In synthesis, there was a marked consistency of predictions with the three models; also, it appears that the temperate forests would be affected the most. In particular, the crucial conifer forests would tend to disappear; second in negative effect would be the cloud forests with reductions of 45–75%; the conifer-broadleaf forests would be reduced by about 35–45%. In contrast, the tropical forests of warm climates would be the least affected. While tropical dry forests would remain about the same, the tropical rain forests are predicted to increase in their coverage under the three scenarios. So, while temperate forests would be drastically

Table 3. The vulnerability of Mexican forests to predicted climatic changes for three model scenarios assuming a doubling of CO_2. Data correspond to percent coverage with respect to total area of the country (~2 million km²). Forest coverage responses are indicated as positive (+), negative (-), relatively insensitive (~), or complete disappearance (0). Modified from Villers and Trejo (1998).

Forest type	Current potential	CCCM scenario	GFDL-R30 scenario	+2°C,-10% rain scenario
Evergreen tropical	5.86	6.67(+)	7.85(+)	6.40(+)
Deciduous tropical	24.28	25.22(+)	28.67(+)	24.74(~)
Montane cloud	2.10	0.54(-)	1.30(-)	0.26(-)
Conifer-broadleaf	6.36	3.65(-)	3.92(-)	3.91(-)
Conifer (cool)	2.31	0	0	0

reduced, tropical forests would not be affected or would even increase. This potential for coverage expansion assumes, of course, that forests will be able to track such predicted climatic changes. However, forests of all kinds face the other major threat, land use (discussed above), that might compensate for the potentially positive effects, or exacerbate the expected negative effects, of climate change.

Consequences on Ecosystem Functioning and Services

Besides the loss of species and populations, human activities also constitute a significant threat to the functioning of forests and the services that they provide to human society as a whole. Here I review, by way of examples, two types of consequences, one dealing with carbon emissions due to deforestation, and one dealing with the disruption of ecological interactions due to forest fragmentation and overexploitation.

Deforestation and carbon emissions

Masera et al. (1997) have calculated total carbon emissions due to deforestation, and the relative contribution of deforestation by forest type and forest conversion activity. This is the most comprehensive analysis available of this ecosystem process for Mexican forests. Total carbon balance was estimated in 53 Megatons of carbon released to the atmosphere per year in the mid 1980s, due to land use. This figure represents about 40% of the country's total carbon emissions. More important in the context of biodiversity at the level of ecosystem types, the study showed that the greatest contribution to carbon emissions is due to tropical forest deforestation. In particular, tropical evergreen forest deforestation contributed with 50% of the emissions, followed by an additional 33% by tropical dry forests. Thus, tropical forests, and particularly tropical evergreen forests are the most critical for this ecosystem service. In addition, this study included an analysis of the contribution to carbon emissions as a function of types of land use. Such analysis indicated that, regardless of the type of forest, the type of conversion that contributed the most to carbon emissions was conversion to grasslands for cattle. Once again, this type

of land use is the least compatible with ecosystem services, in this case related to sequestration of greenhouse gases.

Land use and biotic interactions

The maintenance of biotic interactions is another important ecosystem function. Here I refer to the effects of forest fragmentation and over-exploitation on the disruption of plant-animal interactions. Animals are an important factor in the ecology and evolution of plants, because they affect several processes and stages of a plant's life cycle. Yet the role that medium- to large-size mammalian herbivores such as agouties, peccaries, deer, or tapir play in the structure, functioning, and diversity of tropical forests is very poorly known. Contemporary hunting, over-exploitation and deforestation/forest fragmentation are causing a significant decline in abundance or outright local eradication of several species of this group of animals in tropical forests in general (Redford, 1992), and in some Mexican tropical forests in particular (Dirzo & Miranda, 1991). Comparisons of forests with contrasting degrees of conservation of their mammalian fauna provide some insights on the consequences of the loss of mammalian herbivores. Dirzo and Miranda (1991) carried out a comparative analysis of the levels of mammalian herbivory in the forest understory of defaunated and intact sites. Their results indicated that intact sites experience a load of herbivory in which about 30% of the plants and leaves are eaten by mammals, in comparison to zero in the defaunated forest. Such defoliation is accompanied by seed predation and trampling of seedlings of the forest understory. Associated with this, a comparison of the structure and diversity of the forest understory showed significant changes associated with the presence or absence of the fauna. In summary, it appears that defaunation leads to floristically impoverished, monotonous carpets of seedlings in the defaunated forest. Given that the understory seedlings constitute the regenerative phase of the mature trees that compose the forest, defaunation may affect the most important characteristic and service of tropical forests: their incredible biological diversity and provision of resources for human societies.

These examples of the ecological consequences of the predominant threats for Mexican forests highlight the fact that the numerous functions, services, and potential economic values of such forests have not been considered. A total economic valuation of Mexican forests should be a useful, first-step instrument for the recognition and accounting of the value of forests to society.

Total Economic Valuation of Mexican Forests

As Pearce (1998) rightly pointed out, the current global crisis of forestry stems from the fact that non-timber functions and services of forests (maintenance of biological diversity, carbon sequestration, life-support functions, non-timber products, recreational use, passive value) are becoming increasingly scarce. The expectation that such a scarcity, which should raise the price of non-timber functions (thus

encouraging new supply), does not happen that way, is due to two fundamental problems (Pearce, 1998). First, local market agents involved in the generation of these services cannot compete with the more powerful markets of the forest concessionaires (e.g., government and multinationals). Essentially, in the rural communities of the developing world the insecurity of property rights leads to uneven competition. The second problem is that many of the non-timber functions simply have no market. Against this situation, the obvious policy involves two steps. First, the definition and demonstration of the value of the non-timber functions, comparing it with the alternative land-use values. If this comparison is favorable to the non-timber functions, the second step consists of the capture of the economic value. This should be based on the consolidation of property rights and the development of competitive markets.

Available information to develop a total economic valuation of Mexican ecosystems is very limited (Adler et al., 1995; Barbier & Burgess, 1996). Nevertheless, the recent exercise of total valuation of Mexican forests carried out by Adler et al. (1995) shows the potential for the application of economic techniques, and tackles the first of the two steps of the model of economic valuation of Pearce (1998) described above. In their analysis they considered the total economic value as the result of the following components: (i) Direct values, specifically the potential returns of tourism and recreation. The use value of non-timber forest products, a potentially important direct value, was not considered due to lack of information and limitations to extrapolate from local studies. (ii) Indirect or functional values, particularly the sequestration of carbon and soil erosion protection by forests. (iii) Option values, i.e., the potential value of resources on the basis of the options they represent, using the case of pharmaceuticals derived from plants of Mexican forests. (iv) The existence values, for which they used, as empirical measures, information on monetary contributions to conservation organizations and programs, debt-for-nature swaps, and public surveys on willingness to pay for the conservation of natural areas.

The total valuation resulting from this analysis is likely to be an underestimate, if only because the value of the non-timber forest products was not considered. The information on the case studies for tropical evergreen and deciduous forests, together with the great diversity of species and products present in Mexican forests (including some 2000 species of ethnobotanical relevance), suggest that the use value of non-timber forest products may be very high, compared to other direct values. Nevertheless, the total valuation yielded a still impressive annual lower bound value of the services of Mexican forests of about USD 4.2 billion. Such values stem from the non-marketed services provided by non-consumptive use; from future potential uses of the genetic resources and from pure existence values; but the largest proportion of economic value comes from the functional values of hydrological and carbon cycling. These estimates suggest that the value of non-timber functions is greater than that derived from the alternative, predominant land uses (Adler et al., 1995; Masera et al., 1997), which are the proximate causes of forest conversion. (See Bromley, 1998, for a discussion of the proximal and ultimate causes of deforestation.)

This analysis, together with my previous descriptions of the biological characteristics of Mexican forests, underscores their nature as multi-functional ecosystems with numerous uses and benefits to local people, *as well as* to more distant users. Also, it becomes apparent that the threats to forests that I described above stem from a failure to recognize and account for these multiple uses and values. Thus, in order to capture at least part of the global values forests provide, national *and* international attention is required to capture their total economic value. Also, it is clear that these values should be incorporated into decisions on the future management of this important resource. In consequence, a final comment on management may be in order.

Toward Forest Management

A detailed discussion of forest management is beyond the scope of this paper, thus I would like to refer, briefly, to the diagrammatic model of Figure 1 to highlight some points relevant to this discussion, and to briefly describe some promising examples of forest management in Mexico.

Forest management includes a range of practices that can be found along an axis of seemingly antagonistic extremes. Such extremes are represented by the intact forests, such as the natural protected areas, on the one hand, and the massive commercial exploitation, largely for timber, or the conversion to grasslands for cattle ranching, on the other. The latter emphasizes commercial production with short-term economic revenues, at the expense of forest degradation. Toward the other extreme, besides the natural protected areas, themselves very small, we find practices, better represented in tropical forests of developing countries, which include the home gardens, or the slash-and-burn agriculture with long-term rotational patterns. These practices, although with some potential for conservation of biodiversity and ecosystem services, have a productive and economic capacity restricted largely to local subsistence. In essence, there seems to exist a tension between conservation management and "productive" management. The development of management schemes located toward the center of the gradient can break such a tension. This would demand combining multiple uses and interests of society at the landscape level. I believe we have almost sufficient ecological principles available to promote designs applicable to such scales and multiple emphases. Such management areas can be designed under social and ecological schemes appropriate to the region and forest type. Some cases with potential in this direction in Latin America have been described by Hartshorn (1995), but Mexico has a history of community-based initiatives for forest management which are located close to the intermediate position of the gradient of Figure 1. These deserve a brief comment.

The tropical forests of the Yucatan Peninsula, but particularly those of the state of Quintana Roo, are being managed by the local communities based on the principle of halting forest destruction by making the rational-use forestry resources a secure and attractive source of income for the local population (Galletti, 1998).

Up to 1953 the forests of Quintana Roo, particularly rich in mahogany (*Swietenia macrophylla*) and Spanish cedar (*Cedrela odorota*), were logged without any management plan and in a very chaotic fashion. As from 1953, and for three decades, the forests were logged under a government concession to a parastatal industry (MIQRO), which devised a clear management plan. The plan consisted of a polycyclic system, with a logging cycle accommodated to the 30-year period of the concession. All marketable trees were to be cut in 25 years, with a program of regeneration and forest plantations. Although MIQRO faced several problems, by the end of the concession period there were still important areas of forest left, as well as an important infrastructure, physical (e.g., roads) and managerial (technical training of some of the local inhabitants). At the end of the 30-year concession, federal government policies promoted the participation of the owners in the management and harvesting of their forests. The response to this opportunity emerged as the Plan Piloto Forestal (PPF). The PPF was based on two principles: first, that forests must provide an economic alternative for the local population in order not to be destroyed; second, that forest owners are the social actors most interested in the conservation of the forests, since their principal capital is the forest itself, and not a short-term investment in forest products. To accomplish such a philosophy, landowners first set aside part of their land as permanent forest areas. Next, farmers began to administer the forestry activities. Extraction activities continued using the MIQRO plan, but now under the control of the local farmers. The farmers stopped selling their wood as standing timber and began to develop the capability of processing timber. They also became knowledgeable about calculating production costs and bought their own equipment, eventually developing a commercial union to negotiate the price and conditions of delivery of their products. Despite significant obstacles, important advances toward a more entrepreneurial culture have been made in the local community leading to the development of the local communities and the diversification of forest uses. Remarkable among the latter was the management of chicle (latex from *Manilkara sapota*) for chewing gum production, which was exploited with a philosophy similar to that of the extraction of precious woods. As a consequence, numerous new cooperatives have been organized and the increased profits have led to additional technical advances. Further, institutions that certify sustainably harvested timber for export markets, as well as the development of "green" markets in the U.S.A. and Europe, have helped to develop more ecological criteria for forest management. Subsequently the model of the PPF is beginning to spread in the state, as well as in other parts of the Yucatan Peninsula, such as in the Calakmul Biosphere Reserve in the state of Campeche (Acopa & Boege, 1998).

This description did not address the problems encountered by local forest owners of Quintana Roo. Such problems have been difficult to overcome and some will continue to exist. Nevertheless the societies involved have provided a model of the possible transformation from a forest economy based on the massive exploitation of single forest resources (right-hand side of the gradient in Fig. 2), to one that attempts to move toward the sustainable harvest of multiple products (center of the gradient in Fig. 2) from the rain forest. In these cases, as a highlight, the ecology of forest regen-

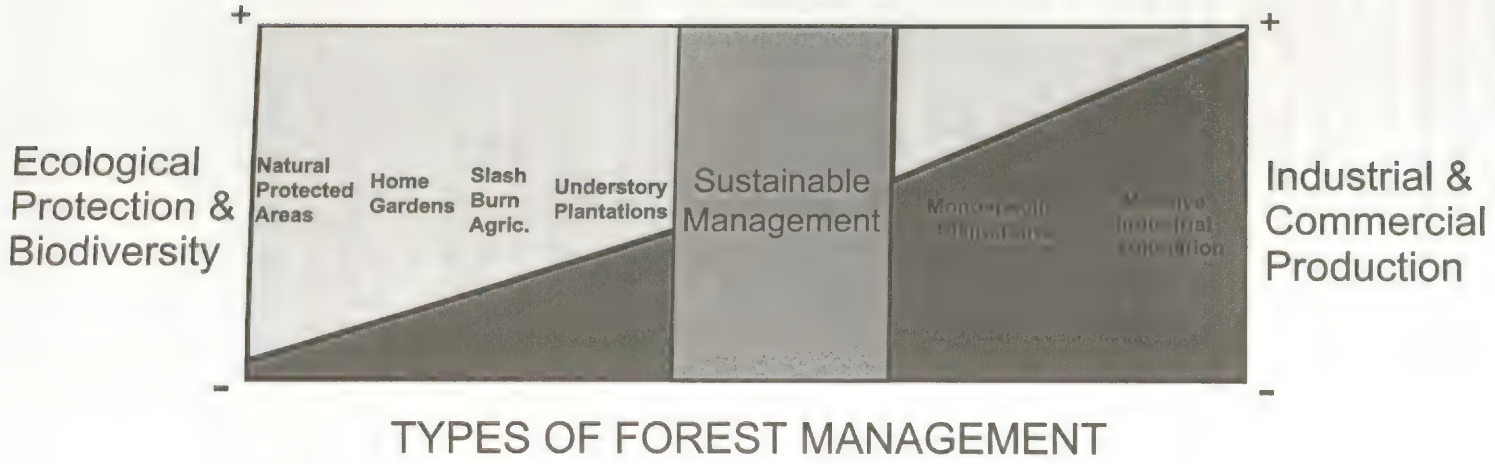


Figure 2. Diagrammatic model representing forest management practices along the biodiversity protection-commercial production gradient.

eration is applied in combination with development of local cooperatives, but efforts are still necessary to locate the forest products in local and regional markets and to capture the economic value of the non-timber functions derived from these forests.

Recently, other local communities have taken the multiple management of their forests in their hands, adapting philosophies and emphases according to their ecological and social characteristics. One noteworthy example is the indigenous community of Ixtlán in the Sierra Norte of the state of Oaxaca (N. A. Montes & G. Ramírez, pers. comm.). The region of this community includes a remarkable variety of ecosystems, from tropical dry forest, to pine and oak forest, alpine prairies, cloud forest, and tropical rain forest. The biological diversity is accompanied by a remarkable diversity of indigenous cultures. Their sustainable management strategy combines traditional agriculture in deforested areas, with selective logging and reforestation in some areas, as well as the promotion of ecotourism directed to the natural forests as well as to the rural communities. The ecotourism project is community-owned so that the profits go to the local people for social and environmental benefits.

Given the increasing realization of the importance of the ecosystem services and resources provided by forests for society beyond the local users, the issue of compensation to local communities (who frequently are forced to exploit forests in non-sustainable manners due to extrinsic economic pressures), should be considered as a crucial component of sustainable rational management.

Some specific considerations to support sustainable management of forests in countries with a profile similar to what I described for Mexico include the following (see Masera et al., 1997, for more details):

1. Efforts to diversify forest production through the development of new markets for the sustainable production of non-precious timbers and non-timber forest products, and to increase the value added of forest products at the harvesting site.
2. To reduce or eliminate subsidies to livestock production and to reduce its comparative advantages against other more ecologically compatible economic activities.
3. To eliminate the disparity between those who own and locally use the forest resources, and those who economically benefit from forests.
4. To incorporate ecological principles into forest management activities and strengthen basic and applied research on forest ecosystem management, as well as to allocate funds for basic and applied research on alternative forest management practices.

Finally, I would like to close with a personal note: an additional important service of forest ecosystems that we have the responsibility to preserve is the inspirational and aesthetic value we derive from them. I am convinced that by taking this into consideration we are helping ourselves in our intrinsic desire to enjoy our planet. Also, given the incredible genetic library we have in forest ecosystems, in so doing we will be maintaining our planet's life-supporting systems, and also, we will be increasing the chances of encountering new valuable resources in the form of wild species/genes that will prove valuable to future generations.

Acknowledgments

I thank Peter Raven for the invitation to participate in the symposium on Human-Dominated Ecosystems, and the suggestion of the theme of this paper. Two anonymous reviewers read a previous draft of this paper and offered extremely valuable and constructive criticisms. Raul I. Martinez helped with several aspects of the preparation of this paper. This paper was written with support from the DGAPA-UNAM and CONACYT (Mexico), during a sabbatical leave at Northern Arizona University.

Literature Cited

Acopa, D. & E. Boege. 1998. The Maya forest in Campeche, Mexico: Experiences in forest management in Calakmul. Pp. 81–97 *in* R. B. Primack, D. Bray, H. A. Galletti & I. Ponciano (editors), Timber, Tourists and Temples. Island Press, Washington, D.C.

Adler, N. W., K. Brown, R. Cervigni & D. Moran. 1995. Total economic value of forests in Mexico. Ambio 24: 286–297.

Barbier, E. B. & J. C. Burgess. 1996. Economic analysis of deforestation in Mexico. Environment and Development Economics 1: 203–240.

Bromley, D. 1998. Deforestation—Institutional causes and solutions. Pp. 95–105 *in* M. Palo & J. Uusivuori (editors), World Forests, Society and Environment. Kluwer, London.

Bye, R. 1995. Ethnobotany of the Mexican tropical dry forests. Pp. 423–438 *in* S. H. Bullock, H. A. Mooney & E. Medina (editors), Seasonally Dry Tropical Forests. Cambridge Univ. Press, Cambridge.

Dirzo, R. 1992. Diversidad florística y estado de conservación de las selvas tropicales de México. Pp. 283–290 *in* J. Sarukhán & R. Dirzo (editors), México Ante los Retos de la Biodiversidad. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), Mexico City.

Dirzo, R. 1994. Mexican Diversity of Flora. CEMEX and Agrupación Sierra Madre, Mexico City.

Dirzo, R. & M. C. Garcia. 1992. Rates of deforestation in Los Tuxtlas, a Neotropical area in southeast Mexico. Conservation Biology 6: 84–90.

Dirzo, R. & A. Miranda. 1991. Altered patterns of herbivory and diversity in the forest understory: A case study of the possible consequences of contemporary defaunation. Pp. 273–287 *in* P. W. Price, T. M. Lewinsohn, W. Fernandes & W. W. Benson (editors), Plant-Animal Interactions: Evolutionary Ecology in Tropical and Temperate Regions. Wiley, New York.

Dixon, R. K., S. Brown, R. A. Houghton, A. M. Solomon, M. C. Trexler & J. Wisniewski. 1994. Carbon pools and flux of global forest ecosystems. Science 263: 185–190.

Galletti, H. A. 1998. The Maya forest of Quintana Roo: Thirteen years of conservation and community development. Pp. 33–46 *in* R. B. Primack, D. Bray, H. A. Galletti & I. Ponciano (editors), Timber, Tourists and Temples. Island Press, Washington, D.C.

Gentry, A. H. 1982. Patterns of Neotropical plant species diversity. Evolutionary Biology 15: 1–84.

Gentry, A. H. 1988. Changes in plant community diversity and floristic composition along environmental and geographical gradients. Annals of the Missouri Botanical Garden 75: 1–34.

Gentry, A. H. 1995. Diversity and floristic composition of Neotropical dry forests. Pp. 146–195 *in* S. H. Bullock, H. A. Mooney & E. Medina (editors), Seasonally Dry Tropical Forests. Cambridge Univ. Press, Cambridge.

Hartshorn, G. S. 1995. Ecological basis for sustainable development in tropical forests. Annual Review of Ecology and Systematics 26: 155–175.

Hughes, R. F. 1997. Effects of Deforestation and Land Use on Biomass, Carbon, and Nutrient Pools in the Los Tuxtlas Region, Mexico. Ph.D. Dissertation, Oregon State University.

Maass, J. M., C. F. Jordan & J. Sarukhán. 1988. Soil erosion and nutrient losses in seasonal tropical agroecosystems under various management techniques. Journal of Applied Ecology 25: 595–605.

McArthur, R. H. & E. O. Wilson. 1967. The Theory of Island Biogeography. Princeton Univ. Press, Princeton.

Martínez, E., C. H. Ramos & F. Chiang. 1994. Lista florística de la Lacandona, Chiapas. Boletín de la Sociedad Botánica de México 54: 99–177.

Masera, O., M. J. Ordóñez & R. Dirzo. 1997. Carbon emissions from Mexican forests: Current situation and long-term scenarios. Climatic Change 35: 265–295.

Mendoza, E. & R. Dirzo. 1999. Deforestation in Lacandonia (Southeast Mexico): Evidence for the declaration of the northernmost tropical hot-spot. Biodiversity and Conservation 8: 1621–1641.

Noble, R. I. & R. Dirzo. 1997. Forests as human-dominated ecosystems. Science 277: 522–525.

Pearce, D. 1998. Can non-market values save the tropical forests? Pp. 255–267 *in* F. B. Goldsmith (editor), Tropical Rain Forest: A Wider Perspective. Chapman & Hall, London.

Potter, C. S., J. T. Randerson, C. B. Field, P. A. Matson, P. M. Vitousek, H. A. Mooney & S. A. Klooster. 1993. Terrestrial ecosystem production: A process model based on global satellite and surface data. Global Biogeochemical Cycles 7: 811–841.

Ramamoorthy, T. P., J. Fa, R. Bye & A. Lot (Editors). 1993. Biological Diversity of Mexico. Oxford Univ. Press, Oxford.

Raven, P. H. 1992. Floristic diversity of the planet and its importance for human well-being. Pp. 81–89 *in* J. Sarukhán & R. Dirzo (editors), Mexico Confronts the Challenges of Biodiversity. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, Mexico City.

Redford, K. H. 1992. The empty forest. BioScience 42: 412–422.

Rzedowski, J. 1978. Vegetación de México. Limusa, Mexico City.

Rzedowski, J. 1993. Diversity and origins of the phanerogamic flora of Mexico. Pp. 129–144 *in* T. P. Ramamoorthy, J. Fa, R. Bye & A. Lot (editors), Biological Diversity of Mexico. Oxford Univ. Press, Oxford.

Toledo, V. M. 1988. La diversidad biológica de México. Ciencia y Desarrollo 81: 18–30.

Trejo, R. I. 1998. Diversidad Floristica de las Selvas Bajas Caducifolias de Mexico. Ph.D. Dissertation, Facultad de Ciencias, Universidad Nacional Autónoma de México, Mexico City.

Villers, L. & R. I. Trejo. 1997. Assessment of the vulnerability of forest ecosystems to climate change in Mexico. Climate Research 9: 87–93.

Whittaker, R. H. & G. E. Likens. 1975. The biosphere and man. Pp. 305–328 *in* H. Leith & R. H. Whittaker (editors), Primary Productivity of the Biosphere. Ecological Studies 14. Springer-Verlag, New York.

Wilson, E. O. 1992. The Diversity of Life. W.W. Norton, New York.

A Note on the Interaction Between Environmental and Economic Factors

Murray Weidenbaum
Mallinckrodt Distinguished University Professor and
Chairman of the Center for the Study of American Business
Washington University
Campus Box 1027, #1 Brookings Drive
St. Louis, Missouri 63130-4899, U.S.A.

Abstract

To a significant degree, environmental improvements depend on the health of the economy (and vice versa). There is a close relationship between poverty and high levels of air and water pollution. When a society reaches an average level of per capita income of about $10,000 a year, it devotes significant resources to improving the physical environment. Two recent economic studies illustrate the potentials as well as difficulties in achieving common ground between economists and environmental researchers. A report by Resources for the Future concluded that the value of the marginal species—using very optimistic assumptions—was only $10,000. However, researchers at the Center for the Study of American Business identified several billion dollars of annual governmental subsidies that were both environmentally harmful and economically wasteful. Although the first study did not result in attracting support from environmentally concerned analysts, the second one did. Public policy would benefit from developing more common ground between the advocates of environmental improvements and economic efficiency.

At the outset of this conference, Peter Raven reminded us that the economy depends on the environment. He, of course, is correct and this paper extends his point by analyzing the other side of that relationship, the extent to which environmental improvements depend on or at least are related to the health of the economy.

Economists do breathe the same air, and we drink the same water as real people. However, we have another way of looking at these phenomena. For example, we find that people tend to waste less water when it is metered. When they have to

pay for their consumption, people are more likely to turn off the tap when they are not using the water and to repair leaks more quickly.

At a more macro level, we find that there is a close relationship between poverty and high levels of air and water pollution. However, this positive relationship does not hold for all environmental indicators. For example, municipal wastes per capita and CO_2 emissions per capita have an inverse relationship to the incidence of poverty (World Bank, 1992: 10–12). Nevertheless, some years ago, economists found that, when a society reached an average level of per capita income of $5,000, it could and did devote significant resources to improving the physical environment (Coursey, 1992). Today, that cutoff may be closer to $10,000 per capita (Grossman & Krueger, 1995).

Any doubters about the relationship between income and pollution should recall the high levels of pollution encountered in Eastern Europe at the time that the Berlin Wall came down. Those who had the occasion to travel from Western Europe to Eastern Europe were struck by the stark differences in economic and pollution levels.

In a world of limited resources, economists are concerned about achieving the most productive uses of those resources. Turning to the United States, the Environmental Protection Agency reports that this nation spends over $150 billion a year for environmental matters. The reaction of many economists is that, with some changes in regulatory approaches, we could achieve the same environmental quality at lower cost or more environmental quality at the same cost—or we could split the difference and make progress on environmental and economic fronts simultaneously (Committee for Economic Development, 1998).

To illustrate the potentials as well as the difficulties of achieving common ground between economists and environmental researchers, I would like to report on two economic studies that relate to environmental matters.

An Economic Analysis of Biodiversity

It seems appropriate to raise a serious question, which may provoke some spirited but hopefully productive dialogue. In that light, it must be reported that, trying as hard as we may, economists have not yet been able to make a compelling *economic* case for biodiversity. It should be emphasized that economists do not conclude that biodiversity is not valuable. Rather, our point is a far more modest one: we still have to depend on ecological, moral, aesthetic, and other values to justify efforts for conservation.

To illustrate this notion, it may be helpful to briefly describe one recent empirical effort along these lines. Resources for the Future (RFF)—the leading U.S. research institute specializing in the economics of the environment and natural resources—assigned a team to put a value on biodiversity as it applies to generating new pharmaceutical products. The medical area surely is one in which we would expect some of the most promising commercial applications of species variety.

Under an extremely optimistic set of assumptions, the RFF economists reluctantly came up with a "maximum possible value" of the marginal species of only $10,000 (Simpson et al., 1997: 177). The assumptions dealt with such factors as the probability of finding a useful medicinal product from the many species to be tested as well as the numerous costs involved. The course from identifying a promising new chemical compound to obtain regulatory approval and on to possible commercial success is well known to be long and costly (U.S. Office of Technology Assessment, 1993). When the RFF team used somewhat less optimistic but quite reasonable assumptions, that produced values even lower than $10,000 per marginal species.

Although the concept of the marginal item may sound esoteric to non-economists, a little explanation may be useful. Most issues are not matters of either/or but of more or less. Thus, in reality, the practical argument usually is not whether or not to reduce the amount of air or water pollution, but how much to reduce it in view of the rising costs of each additional benefit. In this manner, economic analysis tends to operate at the margin.

The RFF economists also attempted to cost out a widely used model in the ecological literature incorporated by MacArthur and Wilson (1967) in their influential theory of island biogeography. Analyzing 18 biodiversity "hot spots" and employing a sophisticated but widely used methodology known as "willingness to pay," the economists estimated the marginal value of a hectare in western Ecuador at only $20. In Tanzania, the market value of land was estimated even lower, at $2 a hectare (Simpson et al., 1996: 176). The use of monetary values here is not a mere "green eyeshade" reference to dollars and cents. Rather, it is the economist's way of analyzing the notion of opportunity cost, which is the sacrifice of other things that we value when attempts are made to protect marginal amounts of land and biodiversity.

To state the matter bluntly, results such as these are not very encouraging. However, they may provide incentive to others to develop new analytical techniques. After all, the RFF economists concluded that they found it impossible to estimate the value of the marginal species "with any precision" (Simpson et al., 1996: 175). Moreover, some analysts contend that issues such as biodiversity raise such strong intergenerational concerns that the current regime of property rights should be reexamined (Norgaard, 1992).

Although economists seem destined to bring a wet blanket to whatever party they are invited to, this note concludes by offering a positive point. A team of researchers at the Center for the Study of American Business at Washington University recently identified government subsidies to mining and other industries that were both economically undesirable and ecologically harmful (e.g., using federal tax or expenditure incentives to artificially pump up the use of fossil fuels). This initial effort came up with several billion dollars of these "two-fers" (Weidenbaum et al., 1997).

To eliminate such items from the federal budget will take a great deal of combined and sustained effort. It will not be easy to offset the entrenched political

power of the special interests defending the status quo. The business reaction so far to the CSAB study has been far from supportive. We certainly have not received the enthusiastic support anticipated for equally objective analyses of subsidies to labor and agriculture.

Public policy in this area would indeed benefit from developing more common ground between the advocates of environmental improvements and economic efficiency. This pioneering conference has provided a splendid opportunity for starting such an advance in the public policy process.

Acknowledgments

The author is indebted to two anonymous reviewers for helpful suggestions.

Literature Cited

Committee for Economic Development. 1998. Modernizing Government Regulation: The Need for Action. CED, New York.

Coursey, D. 1992. The Demand for Environmental Quality. Working Paper. Washington University Olin School of Business, St. Louis.

Grossman, G. M. & A. B. Krueger. 1995. Economic Growth and the Environment. QJE 110: 353–377.

MacArthur, R. H. & E. O. Wilson. 1967. The Theory of Island Biogeography. Princeton Univ. Press, Princeton.

Norgaard, R. B. 1992. Sustainability as intergenerational equity. Environ. Impact Assess. R. 85–124

Simpson, R. D. et al. 1996. Valuing biodiversity for use in pharmaceutical research. JPE 104: 175–177.

U.S. Office of Technology Assessment. 1993. Pharmaceutical R&D: Costs, Risks, and Rewards. U.S. Government Printing Office, Washington, D.C.

Weidenbaum, M. et al. 1997. Toward a Healthier Environment and a Stronger Economy. Washington University Center for the Study of American Business, St. Louis.

World Bank. 1992. World Development Report 1992. Univ. Press, New York.

The Contribution of Economics to Ecosystem Preservation: Far Beyond Monetary Valuation

Thomas Michael Power
Professor of Economics and Chairman of the Economics Department
University of Montana
Missoula, Montana 59812, U.S.A.

Abstract

It is often assumed that the primary contribution that economists can make to environmental analysis is to estimate the monetary value of ecosystem services. While this may be useful in particular situations, such an effort would be incredibly data, time, and resource intensive. In most real-world environmental conflicts, economists cannot reliably generate this type of information, parties to the conflict would not respect or use this type of information, and it is not clear that productive steps at resolving the conflict require this type of information. The contribution that economists can make to helping resolve such conflicts stretch far beyond this type of determinant modeling and monetary quantification. The contribution that economics can make includes an understanding of all of the following: i. the incentive systems that motivate human action; ii. the alternative ways in which human goals can be pursued; iii. the ways in which public policies encourage uneconomic environmental waste; iv. the structuring of decentralized decision-making systems that are efficient and responsible; v. the cost-minimizing path to achieving socially determined environmental objectives; vi. the linkages between ecosystem functioning and the full range of human objectives.

1. Introduction: The Need for Dialogue Between Moral and Social Morons

Economists and biologists usually do not work well together partly because they are professionally predisposed to be more than a little critical of the ideas expressed

by each other on boundary-area issues. Biologists, for instance, tend to be appalled by economists' efforts to "put a price tag" on biological phenomena such as soaring eagles or endangered grizzly bears and their fascination with and faith in commercial markets (Norgaard et al., 1998; Rees, 1998). Economists, on the other hand, see a primitive authoritarianism in natural scientists' pronouncements about how badly markets work or prescriptions as to how to control human-caused damage to biological systems (Anderson & Leal, 1990). Economists are pictured as moral morons and free market ideologues by biologists, and biologists are seen as social morons and statist ideologues by economists. Not exactly a productive basis for collaboration!

It is important for all those concerned about protecting basic ecosystem functioning to understand that the potential contribution that economics can make to environmental protection stretches far beyond their efforts to attach monetary values to various ecosystem services. These efforts at monetizing environmental values have gained considerable attention and generated considerable controversy. Recently Costanza et al. (1997) caused a stir among both ecologists and other economists by estimating the value of the world's ecosystem services and natural capital at US$33 trillion per year. An entire issue of *Ecological Economics* (Volume 25, 1998) was devoted to debate over the appropriateness and accuracy of this estimate. For a more extended discussion of the economic valuation of environmental services, see Daily (1997). Earlier, efforts by economists to estimate the damage done to environmental resources as a result of the Exxon Valdez oil spill generated lengthy court cases and at least two volumes of economic debate. See Hausman (1993) for the Exxon-supported critique of the monetized damages and the National Oceanic and Atmospheric Administration (1993) for Nobel economic laureates Kenneth Arrow and Robert Solow's defense of these techniques.

Economists' interest in harnessing market-like forces in the pursuit of higher-quality environments is not only a reflection of ideological bias and social parochialism. There is a larger and more fundamental concern that draws economists this direction. Below I outline some of the primary tools (beyond monetary valuation and the extension of markets to new social realms) that economists can bring to bear in the search for solutions to the problem of environmental degradation.

2. Incentive Systems Matter

Economists have found it useful to assume that in general people behave in systematic ways guided by their values and the system of incentives they face. Rather than simply condemning behavior that appears to be destructive and calling for the use of the police powers of the state to eliminate that behavior, economists think that it is productive to try to understand what is motivating people to behave as they do. To the extent that the structure of incentives is rewarding destructive behavior, it probably is wise from a public policy point of view to modify those incentives to bring them at least to neutral if not reversing them so that they en-

courage environmentally responsible behavior (Barbier et al., 1994: 69–77). Understanding the complex set of incentives that legitimately motivates individuals and groups is important in the development of any effective environmental policy. Economists are likely to need the assistance of other social scientists in understanding these incentive systems since monetary incentives are not the only powerful incentives leading people to behave as they do.

3. People and Societies Can Be Flexible and Adaptable

Natural scientists when looking at economic systems see a complex process of physical flows that have to obey natural constraints. Economists looking at the same economic system primarily see the creation of value thorough the satisfaction of human preferences, a process that does not have any specific material component to it (Barbier et al., 1994: chapter 3). Natural scientists tend to take a "requirements" approach to underlying economic relationships (e.g., water, energy, and nutritional requirements), while economists tend to reject that view. Economists are not simply ignorant of the basic biological, chemical, and physical laws. Rather, they are focused upon the fact that human societies have proven themselves to be incredibly adaptable when faced with scarcity and physical constraints. Studying the alternative ways in which the satisfaction of human needs and desires can be pursued allows the development of less damaging alternative production methods and significantly reduces the actual economic cost associated with environmental protection. Resource substitution, technological change, and improvements in social organization offer alternative ways of coping with scarcity and environmental damage (Pearce & Atkinson, 1993; Barbier et al., 1994: 77–82). It is important to fully understand the range of these alternatives as well as their expected limits.

4. Eliminating Environmental Waste

Environmental destruction is not always, or even usually, driven by high-priority human needs. Environmental destruction often occurs in the pursuit of economic values that are trivial or non-existent. Good examples are the huge American federal water projects that are used to grow low-valued crops like alfalfa or potatoes in deserts while displacing similar amounts of agricultural production in other areas where subsidized irrigation is not required (Howe & Easter, 1971). The net result is massive economic loss *and* environmental disruption. Many other examples can be given where the damage to the environment is almost gratuitous in the sense of contributing little to net economic production (Lovins & Lovins, 1998). The source of this uneconomic environmental destruction often lies in public subsidies, tax laws, special interest privileges, or failure of the legal system to recognize the scarce and valuable character of natural systems (O'Toole, 1988). Economics is

very useful in both documenting the waste involved in these situations and explaining the set of social and legal circumstances that lead to the waste.

5. The Importance of Information and Decentralized Decision-making

Rational use of scarce resources in the satisfaction of human needs and desires requires a huge amount of information about the availability and scarcity of hundreds of resources, the evaluation of thousands of production processes, the intensity of the needs and desires of hundreds of millions of people for tens of thousands of different products, etc. If one adds to this list the need to take into account quite local but very diverse and complicated ecological conditions, the staggering size of "the economic problem" should be apparent. Add the constantly changing nature of many of these basic pieces of economic information and it should be clear that centralized economic decision-making is unlikely to be efficient. It is more likely to be catastrophic (Hayek, 1945). Any rational solution to economic decision-making that is going to minimize resource waste will require a significant amount of decentralized decision-making and sophisticated information systems that allow the coordination of those decisions. It is this extremely complex social problem that has led economists to respect the market mechanism as at least a partial solution to this problem. Regardless of the extent of the role one is willing to assign to markets, economics can be useful in helping to analyze the strength and weaknesses of various systems of coordination of decentralized economic decision-making.

Of course, markets often fail where environmental services are involved. In this situation economists can make a significant contribution through the design of alternative economic institutions that harness some of the power of markets to assure that environmental objectives can be pursued in an effective and efficient manner. The development of tradable permits to emit pollutants is one example of such economic instruments that seek to allow informed decentralized environmental decision-making in the face of market failure.

6. Minimizing the Cost of Achieving Social Environmental Objectives

Even when environmental objectives are set without regard to a weighing of costs and benefits, economists can make a significant contribution by evaluating the alternative ways in which those environmental objectives might be achieved and determining which has the lowest cost in terms of other important social objectives. The Endangered Species Act in the United States does not allow the weighing of benefits and costs when a decision is made about whether to list a species as endangered. The political system has (at least nominally) specified that no spe-

cies is to be allowed to go extinct. Having adopted that objective on moral or cultural grounds, however, is no excuse for pursuing it in an inefficient manner. Some approaches to species protection and recovery are likely to be more effective than others and involve less conflict with the achievement of other important social objectives. This point, of course, is not limited to dramatic environmental problems such as species survival. It applies to virtually all environmental objectives. Clearly any rational environmental policy would involve attaining the pre-specified objectives in the most effective manner. Economics can be used in such a cost-effectiveness analysis even if economics has been rejected as a tool to establish the value of the objectives being pursued.

7. Connecting Biodiversity to Human Concerns: "Ecosystem Services"

Economists can help make the link between human concerns and ecosystem integrity and biodiversity (Kiker & Lynne, 1997; Gilbert & Janssen, 1998). It is not sufficient to simply document particular types of ecosystem disruption with which citizens have no experience. If a link is not made to active human concerns, it is unlikely that effective political or social support for ecosystem preservation will be forthcoming. The concept of "ecosystem services" is a good vehicle for developing the connection between what scientists understand and what citizens are concerned about. Although establishing this link is a prerequisite for expressing the value of ecosystem characteristics in monetary terms, this latter step is probably not necessary for effective ecosystem protection. Political and personal action is not always (or usually) based upon quantified monetary measures of costs and benefits, but it *is* usually based upon an informal but thoughtful weighing of costs and benefits. It is here that economists and ecologists need to work together, describing the gains and losses in terms of impacts of human concern.

Survival is not the only, or even the primary, human social concern. One misleading implication of a "requirements" or "necessities" approach to viewing economic activity is the human motivation scheme it suggests: Humanity is driven by necessity and biological survival is the dominant objective. Yet most human societies have not teetered on the edge of survival, hanging by a thread above the abyss of extinction. Instead they have organized and managed their social existence to operate comfortably removed from immediate threats to biological survival (Sahlins, 1972). What has mattered for most peoples has not been merely staying alive but the manner of living. That is what human culture has been all about. Over the past few centuries, as Europeans extended their colonial and then economic control over most of the globe, many traditional societies have unraveled culturally, socially, and economically. As a result, we have gotten a peculiar view of traditional societies, one of a desperate struggle to survive. This is more indicative of human existence amidst social and cultural collapse than it is indicative of the broad breadth of the human experience.

The reason for emphasizing this is that ecologists often emphasize the ways in which environmental degradation threaten continued human habitation of this planet. Where this is clearly the case, that obviously is an important point to make. But that is not the only or primary way to get people to be concerned about environmental degradation. We do not need ecologists to prove to us that grizzly bears and wolves are crucial for human survival before we can mobilize support for protecting at least minimum viable populations of them in their natural habitats. We as human beings are perfectly capable of feeling very strongly about protecting endangered species just as we are perfectly capable of loving our families, immersing ourselves in art, or having a passion for play and adventure, even though none of these are directly necessary for our biological survival. Our commitments to such discretionary qualities in our lives lead to enormous expenditures of time and resources. These non-survival qualities have very high value to us that gets expressed in economic terms.

The point is that environmental quality matters to people and they are willing to sacrifice substantial economic resources in the pursuit of it (Power, 1996). It is to these values that we should be appealing, not just to threats of environmental catastrophe that may or may not be currently defensible. It is not primarily fear that drives economic activity. Why then should environmentalists primarily appeal to fear? We should protect biodiversity because it is beautiful, mysterious, awe-inspiring, intellectually satisfying, fun, and challenging. In addition, it is the source of some of our most deeply felt religious impulses. Finally, we have a moral obligation to other forms of life just as we do to each other. These taken together are more powerful than fear. They are also more human and uplifting. An environmental fundamentalism based on fear has the same liabilities as some religious fundamentalists: a deep distrust of people, an implicit authoritarianism, uncritical self-righteousness, and a suffocating intolerance. That is not the basis for either good science or a good society.

8. Conclusion

In some discussions between economists and ecologists, it has been suggested that the most productive contribution that economists can make is to estimate the "marginal values of ecosystem services" and to specify the "production and demand functions for ecosystem services." While this may be useful in particular situations, such an effort would be incredibly data, time, and resource intensive. In most real-world environmental conflicts, economists cannot reliably generate this type of information, parties to the conflict would not respect or use this type of information, and it is not clear that productive steps at resolving the conflict require this type of information. The contribution that economists can make to helping resolve such conflicts stretch far beyond this type of determinant modeling and monetary quantification. The contribution that economics can make includes an understanding of all of the following:

i. the incentive systems that motivate human action;
ii. the alternative ways in which human goals can be pursued;
iii. the ways in which public policies encourage uneconomic environmental waste;
iv. the structuring of decentralized decision-making systems that are efficient and responsible;
v. the cost-minimizing path to achieving socially determined environmental objectives;
vi. the linkages between ecosystem functioning and the full range of human objectives.

It is important as we seek to encourage dialogue between ecosystem scientists and economists to keep in mind the full menu of potential contributions that economics can make to protecting ecosystem integrity. The emphasis on estimating monetary values for ecosystem elements and the use of the market mechanisms to solve environmental problems is far too narrow and risks crippling or marginalizing economists as productive participants in the next century's most important task: stabilizing and then reducing the level of damage humanity is doing to natural systems.

Literature Cited

Anderson, T. L. & D. R. Leal. 1990. Free Market Environmentalism. Pacific Research Institute for Public Policy, San Francisco, California.

Barbier, E. B., J. C. Burgess & C. Folke. 1994. Paradise Lost? The Ecological Economics of Biodiversity. Earthscan Publications, London.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. J. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260. Reprinted in Ecological Economics 25(1998): 3–15.

Daily, G. C. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Gilbert, A. J. & R. Janssen. 1998. Use of environmental functions to communicate the values of a mangrove ecosystem under different management regimes. Ecological Economics 25: 323–346.

Hayek, F. A. 1945. The use of knowledge in society. American Economic Review 35: 519–530.

Hausman, J. A. 1993. Contingent Valuation: A Critical Assessment. North-Holland, New York.

Howe, C. W. & W. K. Easter. 1971. Interbasin Transfers of Water. Johns Hopkins Univ. Press, Baltimore, Maryland.

Kiker, C. F. & G. D. Lynne. 1997. Wetland values and valuing wetlands. Chapter 10 *in* C. L. Coultas & Y.-P. Hsieh (editors), Ecology and Management of Tidal Marshes: A Model from the Gulf of Mexico. St. Lucie Press, Delray Beach, Florida.

Lovins, A. & L. H. Lovins. 1998. Climate: Making Sense and Making Money. Rocky Mountain Institute, Snomass, Colorado.

National Oceanic and Atmospheric Administration. 1993. Report of the NOAA Panel on Contingent Valuation. Federal Register 59(5): 1062–1191.

Norgaard, R. B., C. Bode & Values Reading Group. 1998. Next, the value of God, and other reactions. Ecological Economics 25: 37–39.

O'Toole, R. 1988. Reforming the Forest Service. Island Press, Washington, D.C.

Pearce, D. & G. Atkinson. 1993. Capital Theory and the measurement of sustainable development: An indicator of weak sustainability. Ecological Economics 8: 103–108.

Power, T. M. 1996. Environmental Protection and Economic Well-being: The Economic Pursuit of Quality. M. E. Sharpe Publishers, Armonk, New York.

Rees, W. E. 1998. How should a parasite value its host? Ecological Economics 25: 49–52.

Sahlins, M. 1972. Stone Age Economics. Aldine-Atherton, Chicago.

Integrating Ecosystem Services into Environmental Law

James Salzman
Associate Professor, Washington College of Law
American University
4801 Massachusetts Ave., NW
Washington, D.C. 20016, U.S.A.
salzman@american.edu

Abstract

Recent research by economists and ecologists on the value of ecosystem services has received widespread attention. To improve environmental protection meaningfully, however, their work will need to be translated into a legally useful form. Integrating ecosystem services into environmental law requires greater research focus on the provision of services, identification of robust indicators of ecosystem service delivery, and ultimately a better understanding of the connection between discrete actions and specific harms to services. In turn, the law can create an information market, driving the demand for research and data on ecosystem services.

With the recent publication of *Nature's Services* (Daily et al., 1997) and a cover article in *Nature* (Costanza et al., 1997), "ecosystem services" have become hot news. Stories on the importance of the services provided by ecosystems have been featured in *Newsweek* and the *New York Times*, on radio talk shows, and even in a segment on television's *Nightline*. These services include maintaining the gaseous composition of the atmosphere (in turn helping to stabilize the climate); running the hydrologic cycle that brings us freshwater; generating, preserving, and making fertile the soils upon which farming and forestry depend; controlling the vast majority of pests capable of destroying crops; pollinating crops; and detoxifying waste.

The work of Daily, Costanza, and their colleagues presents some of the very first rigorous attempts to identify the range of ecosystem services and to value objectively these services in dollars. Their basic message has been both undisputed

and newsworthy: ecosystem services are extremely valuable in economic terms, on the order of trillions of dollars.

While this research has established the range of ecosystem services and their great significance, it is important to take the next steps and consider how these services should be explicitly considered in real-life decisions. For this research to make a difference on the ground, its translation to a legally useful form is critical. Indeed, the concept of ecosystem services holds great promise for transforming environmental law and policy. In crafting their research, scientists need to consider carefully the practical implications of their work in identifying and valuing services.

Perhaps surprisingly, at the moment ecosystem services are rarely included in agency cost-benefit analyses or policy debates (Salzman, 1997). We explicitly value and place dollar figures on "ecosystem goods" such as timber and fish. Yet the services underpinning these goods such as soil fertility almost without exception have no market value—not because they are worthless but, rather, because there is no market to capture and express their monetary value directly.

Replacement costs offer the clearest method to provide dollar values for ecosystem services. Thus when New York City was faced with drinking water that failed EPA standards, it chose to invest $1 billion in natural restoration of the Catskills Mountains watershed rather than construction of a $6–8 billion water treatment plant (Passell, 1997; Chichilnisky & Heal, 1998). New York's decision to restore the Catskills watershed—to invest in natural rather than physical capital for water purification—provides an elegant example of incorporating ecosystem service values into policy. This example, however, is the exception rather than the rule.

Unfortunately, ecosystem services are rarely so easily identified or valued at a *local* scale. Indeed the greatest shortcoming of publications to date on ecosystem services, and openly admitted by their authors, is the macro-scale of the analysis. The fact that soil fertility in the U.S. is worth an estimated $45 billion annually is important to know for general policy direction, but of no help in informing specific land-use or pollution-permitting decisions. One clearly cannot divide the $45 billion value of soil fertility by the nation's total agricultural fields to determine the value of a specific five-acre field threatened by development.

The vast majority of environmental policy decisions are incremental. Faced with local situations, informed decision-makers need to know where and by how much ecosystem services are degraded by specific impacts of development, other land uses, or pollution. This problem, the assessment and valuation of services at the margin, is at once the most important and most difficult challenge for economists and ecologists.

Analysis of services provided by ecosystems has proceeded slowly because ecosystem-level experiments are costly and lengthy (Mooney & Ehrlich, 1997). Moreover, research to date has focused much more on understanding ecosystem processes than on measuring ecosystem services; yet how an ecosystem works is *not* the same as the services it provides. This focus has been reinforced, and partly driven, by regulatory actions. In assessing the adequacy of wetlands mitigation (i.e., restor-

ing or creating a wetland as compensation for dredging and filling an existing wetland), federal and state officials traditionally have compared the sites' *physical characteristics* (e.g., as measured by vegetation or acreage) rather than the services actually provided and their benefits to humans (King, 1997; Salzman et al., 1999). For this reason alone the recent publications are valuable because they raise awareness of the importance in broadening the focus of ecological research to include *provision* of services.

Ecosystem services have real value yet are not understood well enough to be given monetary value. Perhaps surprisingly, current environmental law can help resolve this dilemma through the creation of information markets. For example, our understanding of groundwater chemistry and hydrology has increased tremendously in recent years, primarily because markets for this information were created by enactment of the Superfund law. Potentially responsible parties require a sophisticated understanding of local contaminated groundwater conditions to design the most cost-effective remediation strategies. Their need for information, in effect, creates a market for basic scientific data that rewards increasingly sophisticated hydrology models. Now-wealthy consulting businesses have arisen to create these models and drive the science of groundwater hydrology. Similarly, Clean Water Act regulations have created information markets for data on wetlands vegetation. Indeed, the role of regulation in creating information markets is an important pillar of economics of information theory (Stigler, 1961).

This same logic could create an information market for ecosystem services. As in the examples above, *current* regulations grant authority to demand information on ecosystem services in the context of wetlands restoration and permit assessments (Veltman, 1995; CFR, 1996a), calculation of natural resource damages (Federal Register, 1996; CFR, 1996b), and environmental impact statements (CFR, 1996c). Were such data explicitly required by government officials in permit and damage assessments, as the law already provides, then an information market on ecosystem services data would likely develop. In the context of natural resource damages this would, in turn, spur further research in restoration ecology directed at provision of services.

What does research on ecosystem services offer in return to environmental law? One intriguing possibility is specificity of indicators. Although ecosystem management has become a catchword in government, the most effective *legal* standards to ensure ecosystem protection remain indicator species, such as the Spotted Owl, because they provide specific measures of success (Houck, 1997). Robust quantitative indicators of ecosystem services could provide a complementary legal standard on which to base ecosystem-management strategies. Indicators assessing water flow through wetlands might, for example, include dynamic measures of water retention, nutrient trapping, or water quality. These indicators could be used to mandate local management of ecosystems based on operational standards (i.e., maintaining conditions suitable for provision of baseline levels of services). The direct benefit to people of such conservation actions would be more obvious than the current focus on indicator species.

There are, however, significant challenges to incorporating ecosystem services into environmental law. To bring a private nuisance action for economic loss, one needs to show three things: economic/physical harm, an action that caused the harm, and a causal connection between the action and the harm. The first challenge is showing what is known as a "cognizable injury." In general, this means an injury to you or your property. For most ecosystem services, this is difficult to do. More often, the service's degradation will lead to diffuse public harms (e.g., climate change, loss of water quality), not a specific harm unique to you. And, with rare exception in the case law, generalized harms do not provide sufficient particularized injury to establish standing before the court.

Second, even if you do suffer a cognizable injury, to win a nuisance action you need to show that the challenged act (e.g., the action that damaged delivery of the service) was causally linked to your harm. Because the activities providing ecosystem services are so complex and interlinked, it is very difficult to prove that a specific harm (e.g., flooded land) was causally connected to a specific action (e.g., draining a specific wetland). If everyone's actions contributed to your harm, then practically speaking you won't be able to sue anyone (this is the problem of multiple causation).

Despite these challenges, the role of scientists in developing environmental law should not be underestimated. The recent history of toxic tort litigation, recounted well in the bestseller, *A Civil Action* (Harr, 1995), serves as an important reminder that increased scientific understanding has permitted legal actions that would have had little or no chance of success just 20 years ago. In the 1970s, the door to legal challenges against environmentally harmful actions was often shut because proximate causation—the showing that a specific act likely led to a specific result—could not be shown. Developments in hydrology and toxicology have since established a firmer basis to establish proximate causation in cases of toxic exposure.

It is too early to assess whether scientists' research on the provision of ecosystem services will follow the same path, opening the door to legal challenges against environmentally harmful actions now immune through lack of proximate causation, but the similarities are striking. If future scientific research can credibly connect *specific* actions with the reduced delivery of services and *particularized* injury, the law can and will step in to seek appropriate compensation for the harm caused. Whether we will reach this advanced level of environmental protection remains an unresolved question. It is certain, though, that scientists hold the answer.

Literature Cited

Chichilnisky, G. & G. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–630.

CFR (Code of Federal Regulations). 1996a. Title 43: §11.10.

CFR (Code of Federal Regulations). 1996b. Title 43: §§11.10, 11.14(nn), 11.14(ii).

CFR (Code of Federal Regulations). 1996c. Title 40, §§1502.16, 1502.22(a).

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. J. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Daily, G., et al. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Federal Register. January 5, 1996. 61: 440.

Harr, J. 1995. A Civil Action. Random House, New York.

Houck, O. 1997. On the law of biodiversity and ecosystem management. Minnesota Law Review 81: 869.

King, D. 1997. Using Ecosystem Assessment Methods in Natural Resource Damage Assessment. National Oceanographic and Atmospheric Administration, Damage Assessment and Restoration Program, Washington, D.C.

Mooney, H. & P. Ehrlich. 1997. Ecosystem Services: A Fragmentary History. *In* G. Daily et al., Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Passell, P. 1997. Economic science. New York Times, March 27: D3.

Salzman, J. 1997. Valuing ecosystem services. Ecology Law Quarterly 24: 887–903.

Salzman, J. et al. 1999. Market valuation models and ecosystem management in making legal and policy choices. EPA Grant R826612-01-0.

Stigler, G. 1961. The economics of information. Journal of Political Economics 69: 213.

Veltman, V. 1995. Banking on the future of wetlands using federal law. Northwestern University Law Review 89: 654.

The Capture and Distribution of Ecosystem Service Values

Valuing Ecosystem Services

David A. Starrett
P.O. Box 150821
Ely, Nevada 89315, U.S.A.
davstarret@aol.com

Abstract

We develop the economist's concept of value as opportunity cost and discuss the ways in which decisions imply values through revealed preference. We distinguish between the ideas of incremental and total value and argue that only the former is of importance in practical decision-making. The relationship between price and value is articulated and related to the distinction between public and private goods. Finally, we enumerate the various ways economists use to determine the value of ecosystem services—these include use of prices, methods of revealed preference, voting schemes and contingent valuation.

In discussing economic valuation, one first needs to be clear on what the "value" is supposed to measure. It is possible to think of value in philosophical terms as the "intrinsic worth" of an object, which need not be related to its worth in an economic context. However, here I will restrict to discussion of measures that an economist could justify as relevant in making economic decisions. That is, value will be equated with the economist's concept of "opportunity cost"—the value of the foregone alternative. For example, in assessing the value of clean air, we will want to measure the amount of other resources society would be willing to devote to improving the quality of air (or preventing its degradation). Or in calculating the value of pollination services we would need to know how many resources would be enough to compensate for their loss.

Clearly in order to get a single measure of value in these circumstances, we need a way to aggregate the compensating resources and a precise statement as to how much air quality and pollination is "at risk." In this paper, we will discuss ways of assessing these kinds of values. We begin with a general discussion of valuation from an economist's perspective and later turn to specific issues that arise in connection with valuing ecosystem services.

I Objectives, Values and Tradeoffs

The central problem in economics is one of allocating scarce resources and the presumption is that our objective is to get the greatest possible "social benefit" from these resources. Most people would not criticize the economists' agenda as just stated. Getting the most from what you have to work with seems like a worthwhile objective. Controversy comes from trying to get specific about the concept of social benefit. For example, if social benefit is equated with gross national product, then the objective becomes quite materialistic and can be criticized on many grounds. Economists frequently are identified with this materialistic view in the popular press, but this criticism is quite unfair. Our discipline has spent considerable time and effort identifying and trying to measure other factors that enter into social benefit. These include such diverse items as: the quality of life, clean air and water, sense of security, life expectancy, availability of parks, and biological diversity. And as we will explain below, an economist's concept of valuation for an item is, roughly speaking, its contribution to social benefit.

a. Necessity of tradeoffs

What does distinguish economists from others who think about social objectives is that we find it necessary to aggregate all these diverse items into a single measure. Put another way, we find it necessary to consider tradeoffs among these items and assign values to them. A simple example will be used to illustrate the considerations here. Suppose we are considering a nuclear power plant that would generate net output worth $10 million, but involves the risk of killing 100,000 people with a probability judged by experts to be one in 100,000. Here the benefit would enter into GNP, and we have implicitly assumed that inputs and outputs can be correctly aggregated using prices (more on this later) so that net benefit is measured in dollars. However, the accident risk is in quite different units and it is not clear how, or even if, it can be aggregated.

All would agree that the necessary comparison is difficult and some would say that we should not try to make it. The economist's position is that we cannot duck the issue even if we want to. A decision must be made. If we go ahead with the project there is a revealed judgement that one "expected" life was worth less than $10 million whereas if we don't, we imply the reverse judgement. Since decisions of this type must be made all the time, we must find an acceptable way to evaluate associated tradeoffs.

One possible resolution is to take the position that human life is priceless and therefore categorically more important than any finite dollar amount. If this position is taken and used to guide decisions, the nuclear project is rejected as is any project that involves a risk of death, no matter how small. This position is philosophically respectable; indeed, there is a significant literature that attempts to identify values in ways unrelated to economic analysis. We do not denigrate this discussion but would argue that it is not appropriate or helpful when making decisions of the sort just discussed.

b. Principles of economic valuation

Most economists believe that two principles should be applied in evaluating tradeoffs: 1. *Consumer Sovereignty* and 2. *Revealed Preference*. According to consumer sovereignty, the tradeoffs used in making social choices should be derived from or at least linked to the tradeoffs held by individual citizens. And according to revealed preference we should, whenever possible, infer people's tradeoffs from choices they actually make rather than what they say they believe in hypothetical situations.

These two principles in conjunction rule out the categorical tradeoff proposed above. Everyone in society takes avoidable life-threatening chances, revealing that they put a finite dollar value on staying alive. For example, the fact that I am willing to drive to the airport on the freeway rather than on city streets (where there would be less chance of being killed) reveals that I am willing to accept a risk of death in order to save a certain amount of time (which has finite value). I will have more to say below about the methodology of revealed preference, and as the reader will see then, it is quite imperfect and not always even helpful. However, it is the economist's position that we have no choice but to do the best we can. The alternative is some form of paternalism whereby some "experts" decide what is best for the rest of us without regard for the latter's preferences. Most of us find that view unacceptable. Having said this, most economists also recognize shortcomings in the use of revealed preference per se. Namely a person's preferences are based partly on information about the world, and if this information is faulty, the resulting preferences will be as well. Thus, we recognize a paternalistic role for experts in providing information about items such as the state of the environment and our current impacts on it.

To an economist, the problem of measuring value is the problem of determining operational tradeoff rates. In particular, value is relative—we have no concept of absolute or intrinsic value. In assessing the value of an object, we ask what is its equivalent in terms of other resources we would be willing to give up in order to have it. The answer, sometimes referred to as *opportunity cost* is operational in that it can be used to determine which changes in social policy are in society's best interests.

II Marginal Valuations

Much of microeconomic analysis focuses on marginal changes and from our viewpoint here, marginal valuations. That is, we ask how to measure the value of a change that is small relative to the size of items such as consumer income. The focus on marginal valuation has both strengths and weaknesses. The major strength is that under fairly weak assumptions (primarily that small changes in one part of the economy induce at most small changes everywhere else), these measures are additive; that is, if we assess the marginal value of clean air and pollination services separately the opportunity cost of a sufficiently small change in both can be

computed approximately by adding measures. From a practical point of view this is important since it allows us to break up the overall valuation problem into simple component parts that are more easily contemplated than the overarching big picture.

However, the major weakness is that changes must be sufficiently small for the approximation to be very good, and it is not always possible to be precise about the goodness of the approximation. And in any event we can be sure that many changes we may be forced to contemplate, such as a nuclear disaster or the melting of polar ice caps, are not small in the sense required. Once non-marginal changes are contemplated, the fact of one change will influence our valuation of another change, and additivity must break down. A simple example will illustrate this point. Suppose we were to ask what is the **total** value of atmospheric services. Since we would not survive if all these services were removed, this total must be more or less all the other resources we have. By the same token the total value of all water services is also pretty much everything. Consequently measures cannot be additive since the total value (opportunity cost) of atmospheric and water services cannot exceed all the resources we have to pay with. The problem here is that once atmospheric services are gone, water resources no longer have value and vice versa.[1]

Primarily for this reason, we would argue that a focus on the economic value of all ecosystem services is not very interesting—any collection of services that are indispensable to human life will give the same intuitively obvious answer. However, it is still important to know how to evaluate non-marginal changes—changes that are not so large as to endanger our existence but are large enough to vitiate the additivity property. We will focus first on ways of measuring marginal willingness to pay and turn later to questions involving non-marginal changes.

a. Use of market prices

As suggested earlier, economists often use market prices as weights in determining relevant tradeoffs. For example, in computing the dollar benefits for our nuclear power project, the economist would weight hours worked by the wage rate and construction materials by the prices of steel, concrete, etc. The logic behind this choice involves an important "theorem" in economics; this theorem tells us that under certain restricted circumstances, allocation of resources through the market system is optimal. From our perspective here, there are two ways to think about this theorem. First, it says that if you let markets run unchecked the resulting prices correctly reflect community preferences so the resulting allocation is "socially optimal." Second, it implies that when market goods (such as steel and concrete) are involved in publicly planned projects (such as the nuclear plant) market prices should be used for measuring marginal social valuation. The basic intuition

[1] It is because non-additivity was ignored that Costanza et. al. [Costanza, R. et al (1997), "The value of the world's ecosystem services and natural capital," *Nature*] derived a number for the value of ecosystem services that exceeded the world GNP (a proxy for total ability to pay). I would argue that their aggregate number is meaningless, although the component estimates might still be useful in some contexts.

here is quite simple—an agent who can purchase multiple units of an item at fixed price will continue buying until the last (marginal) unit purchased has value to him equal to that price. In this way, free access to competitive markets will insure that marginal value is equal to price for all participating agents.

It is critically important to understand the restrictions that must be placed on economic behavior in order that this "market efficiency" proposition hold. Economists agree on the required set of assumptions, but there is a wide range of disagreement as to how closely these assumptions hold in the real world. All agree that the assumptions do not hold universally—indeed if they did, all activity could be left to the market and there would be no need for public intervention in economic policy.

It is widely recognized that some commodities generate effects on third parties that affect welfare but cannot be reflected in the market price. For example, the burning of coal generates air pollution, thus imposing a cost on nearby residents. Therefore, the price of coal will be less than the full social cost of its use, and we must make a correction in order to place the correct social value on coal. We return to a discussion of these corrections in section IV.

Further, there are no markets that fully insulate people from the consequences of a nuclear disaster. And there is a long list of commodities and services for which markets do not exist at all. Indeed, many of the items grouped under the heading of ecosystem services are in this category.

b. Public goods

Economists generally assume that the decision as to whether or not a particular good or service is allocated through the market is made rationally—that is, items are marketed if and only if it is relatively efficient to do so. The biggest impediment to using markets for efficient allocation involve commodities or items that have a "public" character, by which we mean that any individual's decision to consume the item necessarily has repercussions on others. In such situations, the social benefit from individual consumption is necessarily different from the private benefit.[2] As a general rule, we find that such commodities do not have markets. Examples include national defense, public safety, clean air, fish in the ocean, endangered species, and the Ozone layer. All of those items have an extreme "public" character—namely, they must be "consumed" jointly—they cannot be subdivided into "private" chunks. Clean air is such an item: if it is clean for one person, it is automatically clean for everyone. Fish in the ocean have a similar public character: it is not possible to assign any particular fish to any particular consumer so the fish population must be treated as a public resource. Of course, once fish are caught they can be so assigned and take on characteristics amenable to market allocation. Note that other items on the list above have this same public character.

[2] Note that we took for granted that consumption commodities in the previous section did not have this feature, since we implicitly assumed that consumption by one individual affected only his or her own utility function.

Publicness does not entirely preclude the use of markets but does imply that if employed they will work very poorly at best. For example, suppose we try to sell community safety through the market. That is, we allow citizens individually to purchase police time for patrolling the town streets. It is possible that there would be some purchases on this market, but we argue that these will be lower than desirable for the overall social good; when any particular citizen purchases police time, most of the benefits go to others who will benefit equally from the police presence. Here (in contrast to the case of privately divisible goods such as bread or steel) the private benefit from purchase is much lower than the social benefit. Since market price can only reflect the private benefit, the arguments made for efficiency of markets do not hold for goods with these public characteristics. In the economics literature, the problem here is sometimes referred to as one of *externalities*. When one agent purchases services he confers external benefits (or costs) on others who will also be affected. He therefore has what is called a *free rider* incentive to contribute less than would be optimal for the overall social good.

c. The mixed economy and marginal willingness to pay

The simplest paradigm to use in measuring valuation is one in which we divide the list of goods and services into two mutually exclusive and exhaustive categories: those that can be optimally allocated by the market (we label these *private*) and those that are allocated entirely outside the market system through the political process (we label these *public*). Obviously market prices are to be used to reflect marginal valuations of private goods, and the remaining issue in valuation is to determine the tradeoff between each public good and a private goods aggregate, the latter now being measured in dollars using the price weights. Such a measure is referred to as *marginal willingness to pay*.

In the next section, we discuss ways of measuring marginal willingness to pay for ecosystem goods and services in this paradigm. Later, we will address complications that arise when some goods that are marketed are nonetheless not allocated optimally by those markets.

III. Valuing Ecosystem Services

The science (and art!) of measuring economic valuation is quite complicated and generally rather imprecise. Here, we will not try to give a comprehensive treatment of the subject, but rather give a flavor of how things can be done and where the major impediments lie.

In our current paradigm, the first question to ask of a particular ecosystem service is whether or not it is allocated through the market. If yes, then its marginal willingness to pay is given by market price; if not, we must look further. Some of these services are marketed, either directly or indirectly. Agricultural land is marketed directly in most of the world, and rental rates can be used in these cases. These rates should reflect different qualities of the land, including richness, resis-

tance to erosion, nutrient balance, etc., since farmers are willing to pay more for land, the more productive it is in farming. Many raw materials such as metal ores, gems, and other minerals are mined privately and traded on markets. In some cases the rental rate on mine property may be directly observable. However, when the mine owns the land, it must be inferred from the market value of the mine. Here, there are some legitimate questions as to whether some of these markets are "efficient" in the required sense, and we will address these later. At a more indirect level, pollination services are marketed. Some fraction of this effort is generated commercially by private beehives. These are in direct substitution for naturally provided services and the price earned will reflect marginal valuation that can be applied to those natural services.

There are a few other examples of this type, but a large number of ecosystem services possess the publicness qualities that force them outside markets. These would certainly include atmospheric and oceanic services, estuaries, lakes, and wetlands. How are we to determine the correct marginal valuations to use for public goods? This is an old and vexing question with many proposed solutions, none of them entirely satisfactory. There is a long literature on the design of mechanisms that could provide incentives for participants to truthfully reveal preferences and thereby determine values. These typically involve the sending of messages and a schedule of monetary payments as a function of messages sent, with the understanding that the associated allocation decision will be a known function of messages and payments. The best-known success story here is the "second price auction" for allocating a single item. In this scheme, people bid for the item and it is sold to the highest bidder for the *second highest* bid price. This mechanism is efficient in that it induces bidders to bid their true valuations and therefore guarantees that the item is sold to the person who values it most.

Unfortunately however, the class of allocation problems for which efficient mechanisms can be constructed is quite limited and generally do not include allocation of items with pure public characteristics (in particular the auction item discussed above does not have public characteristics). Most practical schemes, such as those where citizens vote on public projects together with the tax increases to pay for them, fail this test, largely because they cannot overcome free-rider incentives. And when such mechanisms do exist, they are generally too complicated to implement from a practical point of view.

Another possibility is to learn something about preferences for public goods from observable behavior, typically behavior on related private goods markets. For example, one can presumably learn something about the value of clean air by observing differences in rents on clean versus dirty sites. Or one can learn about the value of job safety by observing the differences in wages between safe and risky jobs. These procedures of indirect revelation are known as *hedonic methods* and there is a large literature on their theoretical foundations as well as attempts at empirical implementation. Again, the areas of success are limited. The class of public items for which hedonic methods are amenable is limited and even when available, the practitioner generally faces problems of identification. For example,

in the case of clean air, land rents vary across locations for many reasons other than ambient air quality (view, convenient access to shops, work, etc.), and the method works only if one can identify which parts of the rent difference are due to which cause.

When all else fails, there is still the option of conducting surveys in which people are presented with a hypothetical situation involving a change in public goods and asked how much it is worth to them. Such methods have become more and more popular over time and are now grouped under the heading: *contingent valuation*. As long as participants believe the situation described is strictly hypothetical, they have no incentive to lie and one might hope would therefore reveal their true preference (even though they have no incentive to tell the truth either). However, experience with these methods is mixed at best. It is clear that responses are quite sensitive to the way in which questions are asked. Furthermore, research has shown that responses tend to be biased in a number of ways. Part of the difficulty with contingent valuation may be that when questions are strictly hypothetical, respondents do not think very hard about them and consequently do not give the more accurate answers they would if money were on the line. However, as we have suggested above, when money is on the line, incentives generally become distorted, so contingent valuation methods frequently are the only game in town. Much work is currently going on to design contingent valuation methods that will better elicit correct information.

IV. Implications of Market Distortions

The argument for using market prices as proxies for marginal social preference rests on a number of assumptions concerning the workings of markets. When any of these fail to hold, market price must be modified in an appropriate way. Among the various causes of market failure, the two most important for us are externalities and inequities.[3]

In any market, there is the possibility that some benefits (or costs) of the associated activity accrue to people not engaged in the activity. We saw that this happens naturally if we try to allocate public goods through a private market, but it may happen more generally. For example, when coal is burned it emits particulates that dirty the air and greenhouse gasses that may contribute to global warming. If we treated the particulates and gasses as separate commodities, they could in principle be assigned their own (negative) marginal social valuations, but this is rather difficult to do in practice for the same reasons we have difficulties in valuing pure public goods. Otherwise, the price of coal understates the true marginal

[3] Standard treatments of "market failure" would include a discussion of monopoly power. When one firm dominates a market, it has substantial control of its output price and an incentive to raise that price above marginal social cost. Evidence suggests, however, that even if monopoly power were prevalent (debatable) the distortions in market price would be relatively small, so the issue is of minor concern from our current perspective.

social cost of use, and we must add something to reflect the cost of the externality. An implication of this is that such markets ought to be regulated, for example by imposing a tax to reflect the externality cost. Unfortunately, computing the appropriate tax requires the same kind of information as in the previous section and is consequently an inexact science. When, for whatever reason, the correction is not made in the market, then market price alone is not a correct measure of marginal valuation.

Another type of externality is generated by so-called *existence values*. An item has existence value to someone if that person gets satisfaction out of knowing that the object is there whether or not they directly take advantage of its presence. This is distinguished from *use value*, which represents the value they get out of using the item. As an example, consider the case of a national park. I get use value out of making visits to enjoy its attributes directly, and this use value potentially can be recovered and identified through entrance fees. However, if I value having the park there quite apart from any use I make of it, this existence value is a public benefit (note that everyone who cares will get the benefit if we decide to have the park at all) that cannot be captured through market charges. To the extent that existence values are important, we could not rely on the market alone to reveal willingness to pay.

As a third example, consider resources such as cattle and oil. One might think these could be treated like mining operations; that is, we could infer the value of the grazing and extraction rights from profits made by cattle and oil companies. However, in many places this would be wrong because grazing and extraction rights are not allocated privately. For example, in open range land where grazing rights are not rationed, one company's grazing generates externalities by reducing the quality of grazing for others. Consequently, land will be overgrazed and grazing rights will be undervalued at the margin (in the literature, this situation is referred to as the *problem of the common*). In some circumstances the same argument might be made with respect to mining rights, although these are generally more localized and therefore immune to this externality.

Another example will illustrate the controversial nature of this subtopic. Consider the provision of primary education. Some would argue that such education is a private commodity—those doing the learning get the benefits and we should allow private choice in how much and what kind of education to purchase. In their view, a freely functioning private market would generate a price that correctly measures marginal social benefit, and no government interference would be justified. The counter-argument involves externalities to some degree but is mostly driven by issues of inequality. In the market system a person's command over resources is proportional to income, which naturally leads to inequality in outcomes. If primary education were strictly left to the market, people would have correspondingly unequal access. Many would argue that there should be equality here—children should start out in life with "equal opportunity." Consequently, they argue for public provision of primary (and even secondary) education.

The issues here are quite separate from those of efficiency raised earlier[4], and the reader should see that egalitarian beliefs militate against reliance on markets generally; indeed they are at the heart of the debate about capitalism versus socialism. Since the crux of this debate involves a value judgement, there is no right or wrong answer and different societies resolve the conflict between efficiency and equity goals in different ways. In the U.S. economy, a large percentage of allocation decisions are made through the market with varying degrees of regulation through taxes, standards, and the like. At the same time there is a widely (but not universally) held belief that the resulting market outcome is inequitable to some degree. There is in principle a way to modify marginal valuations to account for equity considerations but it is difficult to implement and rarely done in practice.

V. Non-Marginal Changes

The methodology of marginal willingness to pay is powerful but can never be comprehensive because it can never deal effectively with large divergences from the status quo. For example, suppose we wanted to consider a policy of curtailing all energy production from nuclear and coal-powered plants. What price should we use to evaluate the lost energy? There is no obvious answer since the price of energy surely would change (rise) as a consequence of the policy. Further, the policy would cause other prices (or marginal valuations) to change as well and thus have indirect effects throughout other sectors of the economy. Consequently, when large changes are considered, we cannot avoid looking at the big picture.

Eliciting the necessary information for deciding on large changes therefore becomes much more difficult. Without going into detail, we indicate some of the new impediments. (1) With complete information for consumer demands as a function of price and income, it is possible to measure consumer willingness to pay for large changes in market variables.[5] However, the information needed is almost never available, particularly at the individual level, and the best we can do is construct crude approximations. (2) Hedonic methods can sometimes be generalized in similar ways, but the identification problems are much more demanding on the data. (3) The types of questions we would have to ask in contingent valuation surveys now will involve the simultaneous change in many variables, requiring respondents to think about much more complicated hypothetical situations. (4) Non-additivity implies that studies cannot be conducted independently without the danger of

[4] The efficiency "best served" by markets is called *Pareto Efficiency*. An outcome is Pareto efficient if there is no way of making one person better off without making someone else worse off. This concept ignores equity considerations, as it pays no attention to the relative well-being of citizens.

[5] Intuitively, each unit of the item gained (or lost) in the large change is valued at the marginal value it would have if it were the last such unit and the incremental willingness to pay (or be paid) is computed as the sum of these values. Such measures are sometimes referred to as *consumer surplus* or *compensating variation*.

making mistakes. For example, we may find that two changes are desirable separately but not jointly. This means that studies need to be done in sequence, and the correct sequence is not identifiable at the outset.

Faced with these and more daunting impediments, the best we can do is rely on certain kinds of approximations. Among the most useful of these are upper and lower bounds. For example, in assessing the costs of a nuclear disaster, we can get a lower bound by adding up the expected hospital and funeral costs from those who become sick or die, and the expected cleanup costs at the plant, at fixed prices for the services. This will surely understate the full cost, but if it already exceeded the expected net benefits of the nuclear power we would need no further information and should abandon the project. Similarly, we could use the market cost of coal as a lower bound on its true social cost, assuming only that we would be sure that all its externalities were negative. Such methods can be really powerful in helping us to accept the "really good" changes and reject the "really bad" ones. And, of course, getting those decisions right is extremely important. Unfortunately, for changes in between our current methodology is not powerful enough to give definitive answers.

Annotated Bibliography

The survey of valuation topics given in this paper merely skims the surface of a very large subject. The reader should consult textbook/handbook style references below for more in-depth coverage of indicated topics.

Arrow, K., R. Solow, P. Portney, E. Leamer, R. Radner & H. Schuman, "Report of the NOAA Panel on Contingent Valuation," Federal Register 58: 4602–4614. This report discusses the various strengths and weaknesses in Contingent Valuation methods with special emphasis on studies done in connection with the Exxon Valdez oil spill.

Baumol, W. & W. Oates. 1988. The Theory of Environmental Policy. Cambridge Univ. Press, New York. This book goes into considerable detail on the theory of collective goods and externalities and the institutional arrangements that have been created to deal with them.

Dasgupta, P. & K-G. Maler. 1992. The Economics of Transnational Commons. Clarendon Press, Oxford. The authors discuss management of common property in situations where the relevant commons extends across political constituencies.

Davis, D. & C. Holt. 1993. Experimental Economics. Princeton Univ. Press, Princeton, New Jersey. This book looks at the ways in which economic actors deal with economic situations such as the free rider problem in the setting of laboratory experiments.

Freeman III, M. 1979. The Benefits of Environmental Improvement. Johns Hopkins Press, Baltimore. This work looks at the various ways of measuring environmental improvements, including hedonic methods, use value assessment and travel cost methods.

Handbook of Public Economics, Vols.1, 2. 1987. Auerbach, A. & M. Feldstein (eds.), North Holland, Amsterdam. This reference work contains a survey of topics in the areas of Public finance, political economy and the theory of collective goods.

Hausman, J. (ed.). 1993. Contingent Valuation: A Critical Appraisal. North Holland, Amsterdam. A variety of opinions are expressed concerning the methodology of contingent valuation and their efficacy in revealing private valuations.

Inman, R. "Markets, Government, and the 'New' Political Economy." Chapter 12 *in* Handbook for Public Economics, Vol.2, Op. Cit. Managing collective goods using methods and institutions of group decision-making are discussed and assessed.

Laffont, J. 1987. "Incentives and the allocation of public goods." *In* Handbook of Public Economics, Vol.1, Op.Cit. Here there is a discussion of the art of the possible and impossible in constructing mechanisms to solve various economic allocation problems.

Martin, F. 1989/1992. Common-Pool Resources and Collective Action: A Bibliography. Indiana Univ. Press, Bloomington. Here is a compendium of studies, both theoretical and empirical, of ways collectives have (or could) organize to manage common property resources.

Kopp, R. & V. Smith (eds.). 1993. Valuing Natural Assets: The Economics of Natural Resource Damage Assessment, Resources for the Future, Washington, D.C. This compendium discusses ways of measuring externalities associated with environmental damage. Both direct methods (such as contingent valuation) and indirect ones (such as hedonics) are evaluated.

Starrett, D. 1988. Foundations of Public Economics. Cambridge Univ. Press, New York. In this monograph are discussions of the theory of collective goods, cost benefit analysis, and the art of the possible using hedonic methods.

Valuing Forests

Bengt Kriström
Professor of Resource Economics
Department of Forest Economics
901 83, SLU-Umeå, Sweden
and
Economic Council
Ministry of Finance
103 33 Stockholm, Sweden

Abstract

Forests play an important role for human well-being in many dimensions. Forest products such as paper and pulp remain important goods on the world markets, as do many other marketed products directly linked to forest resources. According to the FAO, more than 50% of the world's population is dependent on forests for daily energy needs, such as cooking. Employment opportunities may be of significant importance locally, regionally, and nationally. In addition, forest ecosystems provide many "ecoservices" that contribute to human well-being, even though those services do not necessarily fetch any market price. In recent years, resource economists have begun the task of quantifying the contribution of forest resources to human well-being. There are comprehensive forest valuation studies in more than 100 countries. Valuation studies have uncovered the significance of forest resources and provided a deeper understanding of the many ways in which forest resources benefit mankind. This paper provides some insight into these efforts. It begins by addressing the fundamental question: what is it that we want to value? This question has a clear answer from an economic point of view, yet no consensus exists in the relevant literature, in particular when trying to augment the value of forests in expanded ("green") national accounts. The second part of the paper briefly discusses strengths and weaknesses with currently available measurement methods, especially with regard to the measurement of the non-market outputs from forests. Thus, this part of the paper sheds light on how we can value forest resources. The third part of the paper deals with the practical question: how have we valued our forests? It presents some examples of how valuation studies at dif-

ferent levels of aggregation can yield important insights into a farsighted and balanced husbandry of forest resources. The final section has concluding comments on the usefulness of valuation in this context.

Introduction

Forests have served mankind since time immemorial, and they continue to play a significant role for human well-being. While most of the attention from economists used to be on the market value of (certain) forest products, non-market values of forests are now increasingly being appreciated and measured. A significant number of studies on non-market values of forest have been carried out worldwide (a recent summary appears in Forestry Commission, 1999). This reflects an evolving research agenda that attempts to broaden our understanding of the linkages between the economic and ecological systems. To some observers, this development is as extraordinary as it is pleasing. To others, the current focus on the non-market value of forests is misguided, given the paucity of data and the purported weaknesses of measurement approaches. Whatever the case may be, it is a good time to take stock and contemplate the input of resource economists to our understanding of forest management. To be sure, the sheer volume of contributions makes any survey incomplete, and this paper is no exception.

I address three questions relating to the valuation of forest services. The first is: What is it we want to measure? This question has been answered in many different ways in the literature. Resource accounting is a particularly diverse field along this dimension. This is unfortunate, because there is no reason for disparate views on certain fundamental principles of valuation. Or so I will argue. The second question is: What is it that we want to value? There are a number of useful methods, still controversial in some cases, that have been significantly improved in recent years. The usefulness of these methods vary across applications. I illustrate this fact by means of sequence of examples organized by the level of aggregation. I begin at the "top level" (the value of all forest ecosystems in the world), proceeding to the national level (the "true" contribution of forest ecosystems to the national product), and end with studies on the household level. In the concluding section of the paper, I discuss the third question: Is valuation helpful in the quest for a proper husbandry of forest resources? I conclude that valuation is useful, if we keep the valuation studies within a tightly specified theoretical structure.

What Is It We Want to Measure?

In an earlier conference between economists and ecologists, Walter Heller, an important economist in the 1960s presidential administrations, opened the conference with a declaration of humility. He noted that although we know a lot theo-

retically about the complex interplay between the economy and the environment, we know little empirically. This was in 1971. We have now learned a little more, not the least theoretically. The relevant economic theories often require as an input the value of unpriced ecological services. I believe there is wide agreement today that "all" forest services and goods should be empirically measured, even though economic theorists may not be as optimistic as others on the validity of currently available measurement approaches. There also seems to be a certain degree of communality between different stakeholders regarding valuation, as, e.g., reflected in statements by major organizations such as the World Bank: "Second, on biodiversity; we simply won't succeed unless environmental values can be reflected in the marketplace" (remarks to the United Nations General Assembly Special Session on the Environment, by James D. Wolfensohn, President, The World Bank. New York, June 25, 1997). In addition, Principle 6c of the RioPrinciples states, "The development and improvement of methodologies for such evaluations should be promoted."

There is another argument in favor of valuation, which is more fundamental than the fact that something that is not valued is often downplayed or forgotten in the final analysis. As the economist Ciriacy-Wantrup argued in 1955 (Ciriacy-Wantrup, 1986: 136): "There would then [without valuation] be no calculation in which the public could have confidence and that could be used as a standard to measure various claims. Furthermore, there would be no calculation easily open to public scrutiny in every detail." In short, valuation offers a transparency of decision-making that is appealing to many.

A stumbling block for proponents of valuation has been the choice of numeraire, in particular when trying to persuade non-economists about its usefulness. While not essential to the logic of valuation, a monetary metric is often (but not always) used, simply because it is convenient. As ecologist Daily (1997: 10) pointed out, "...economic markets play a dominant role in patterns of human behavior, and the expression of [economic] value—even if imperfect—is a common currency that helps to inform the decision-making process."

While the need for pricing ecological services, as well as the convenience of using a monetary metric, are no longer major issues for debate, there remains, however, the question of what it is we want to value. It may seem surprising that this question needs to be discussed. But as we shall see in the review of empirical studies, many answers exist. The answer is clear-cut from an economic point of view. What we want to measure is the subjective value, or utility, an individual derives from a particular change of resource use. The economist's job is to map such unobservable utility changes into something that can be measured. The scale used is often money, although this is a scale chosen mostly because of its convenience.

It is important to note that values do not exist in the abstract. In the words of Daily (1997: 7–8), "There exists no absolute value of ecosystems waiting to be discovered and revealed to the world by a member of the intellectual community." This is an important point, because we are typically interested in knowing whether some *change* of resource use is for the better or worse. Therefore, economists are

focusing on the marginal value of resource change, rather than the total value. The total value to humanity of usable water or air is infinitely large, because a change of resource use that would do away with either resource is equivalent to a project that would end human life on earth. It follows by the same kind of reasoning that the value of preserving all the forests in the world is of little interest from an economic point of view.

Valuation of market or non-market goods is based, at least in economics, on consumer sovereignty. Valuation is ultimately based on the idea that consumers have well-defined preferences over different states of the world. Economists are interested in trying to reveal these preferences and with the purported short-sightedness and "ecological illiteracy" attributed to many individuals, valuation of ecological services in monetary terms rests on a basis that some will find unappealing. One possible way to motivate valuation is to think about it in terms of referendums. Those are often used to shed light on a population's sentiments about certain issues, as in Switzerland and California, for example. In Sweden referendums have been held on such complex matters as the future of nuclear power and whether or not Sweden should enter the European Union. When an individual is interviewed about his sentiments toward the environment, economists often include an opportunity for the individual to vote on different ways of using a particular resource. Thus, if it is held that people can usefully express their sentiments toward the use of nuclear power and whether or not a country should enter the European Union, one could argue that we can also obtain useful information from surveys about sentiments toward the use of our environments and natural resources.

There is one additional matter about valuation that is important to discuss here. How much people are willing to pay depends on their ability to pay, i.e., their income (or wealth). The result of a valuation study therefore depends on the existing distribution of income. There is no objective way (in economics, at least) in which we can deal with distributional matters, when aggregating individual valuations. A possible approach is to illustrate how different distributional weights will affect the outcome. Kanninen and Kriström (1992) provided an illustration involving the installation of water taps in a Nigerian village. The sum of willingness to pay is found to be insufficient to cover the cost of the project. If willingness to pay expressed by households with "low income" is given a social weight larger than one, the project costs can be covered. Of course, any choice of weights is arbitrary, but it is important to note that by simply summing the benefits and costs the weights are chosen to be equal to one.

The Value of Forests

If we are to properly appreciate the value to society of a forest, we need to include a range of different goods and services. Indeed, forests are an integrated part of life support systems and play a key role in the regulation of the atmosphere and the climate. Forests regulate local hydrological systems, protect soils from erosion,

and reduce variation in water flows. Forests also play an important role as a repository for genetic information.

The importance of forests to the domestic economy varies across the globe. The traditional products, paper and pulp, still remain important goods on the world markets, as do many other marketed products directly linked to forest resources. Bananas, natural rubber, coffee, and cocoa are important examples. Incidentally, non-timber forest products may yield higher returns than any alternative use of the land in the rainforest, as a careful study from Ecuador shows; see Grimes et al. (1994).

Olembo (1995: 105) claimed that 1290 non-timber species in Malaysian forests are used by humans. He further argued that (1995: 106) "In 1985 the value of medicinal products then derived from forests amounted to $20 billion—without even considering the potential of forests as reservoirs of materials that have not been processed and refined." Indeed, the value, potential or realized, of medicinal products in forests is often thought to be substantial. Some notable "success-stories" may be quoted here. Quinine has long been known to be effective against malaria. The bark and leaves of the willow (*Salix*) contain salicylic acid and was used early on to treat fever and rheumatism (aspirin is the modern synthetic version of this "drug"). The rosy periwinkle of Madagascar was discovered in the 1980s to contain several anticancer alkaloids now used for treating Hodgkin's disease and childhood leukemia. These and other "success-stories," notably the anticancer drug Taxol, suggest that the potential value of drugs hidden in the world's forest could be enormous. However, the summary assessment by Pearce (1999), and recent events at leading pharmaceutical firms involved in "ethnobotany" (e.g., Shaman Pharmaceuticals and Merck), suggest that the net benefits of drugs waiting to be discovered in the world's forests are smaller than previously expected. The studies summarized in Pearce (1999) suggest that the value of pharmaceutical products is between USD 0.01 and 20 per hectare per year in tropical forests. According to the *Economist* (2/20/99), Merck spent 10 years trying to extract and develop the active principles from Chinese herbal remedies, but failed. The bottom line seems to be that one should not expect significant net benefits of potential drugs waiting to be discovered in the world's forests, because technological progress (such as automated mass-screening techniques) currently appears to give the artificial products an advantage.

In addition to the various products that may be extracted from the forests, employment opportunities may be of significant importance locally, regionally, and nationally. For example, the collection and processing of the tendu leaf provide part-time employment for up to half a million women in India (FAO, 1993: 35).

There are many other "use-values" derived from forest resources that have not been mentioned, and the most important is recreation (in various forms). However, a list of values would not be complete without adding non-use values. Such values arise to a person that values the pure existence of a certain forest area, quite independent of any personal direct use of it. Existence values were introduced in the economics literature by Krutilla (1967) and have been the subject of many studies

and controversies (see below). According to Krutilla (1967: 781), "There are many persons who obtain satisfaction from mere knowledge that part of the wilderness of North America remains even though they would be appalled by the prospect of being exposed to it." To some extent, Krutilla's argument was a response to an earlier argument made by Friedman (1962). Friedman (1962: 31) suggested, in his well-known essay on capitalism and freedom, that a simple way of deciding whether or not a park should be preserved is to let the market decide. "If the public wants this kind of activity enough to pay for it, private enterprises will have every incentive to provide such a park." Empirical studies often show (see, e.g., Kriström, 1990) that individuals are prepared to sacrifice some of their material standard to preserve a forest for the benefit of current and future generations, and not only for their use. Altruism appears to be an important drive behind such concerns.

The concept of existence value has been controversial, especially in the U.S. The National Oceanic and Atmospheric Administration (NOAA) released a report in 1993 on the proper role of existence values in damage assessment, in particular whether such values could reliably be measured with the contingent valuation method (the answer being a qualified "yes"). To some economists, e.g., Milgrom (1992), existence values based on altruism toward others are devoid of economic content. He argued that an inclusion of existence values in a cost-benefit analysis involves a kind of double counting. This seems not to be the mainstream view among economists, although altruistically motivated values create some important theoretical problems in cost-benefit analysis. See Johansson (1993: 131–133) for a detailed analysis.

One can also attribute an option value to a forest. It may be useful in the future for other purposes than logging. By preserving the forest, the option to use it for future recreational activities is preserved. The concept of option value was introduced in Weisbrod (1964) as a direct response to the argument made by Friedman (1962). The gist of Weisbrod's argument was that the free market device advocated by Friedman would fail to subsume all the relevant benefits to conservation: "To see why, the reader need recognize the existence of people who anticipate purchasing the commodity (visiting the park) at some time in the future, but who, in fact, never will purchase (visit) it. Nevertheless, if these consumers behave as 'economic men' they will be willing to pay something for the option to consume the commodity in the future" (Weisbrod, 1964: 472).

Weisbrod's concept of option value spurred a large amount of literature, in which much effort was spent in making option value rigorous. It is important to note that option value, as it is typically defined in the current literature, can be positive or negative. In effect, the concept is rather difficult to interpret in general, because it is defined as the residual difference between willingness to pay (option price) and expected consumer surplus. In simple cases, such as when a resource can only take on two levels (preserved/not preserved), (supply-side) option value can be given a meaningful economic interpretation, as in the case considered by Bishop (1982). For more general probability distributions, the concept of option value becomes all the more difficult to interpret. See Johansson (1993: 139–142).

There is a related concept called quasi-option value, which in environmental economics has been used to study projects associated with irreversibilities. It is often interpreted in terms of the value of information. By postponing a project (say a hydropower construction) one may be able to learn more about the ecological and other consequences of the project. This benefit should be weighed against the extra costs involved that are incurred due to the postponement (i.e., foregone income). As shown by Fisher (1988), ignoring the prospect of new information can bias decisions toward development. It is empirically quite difficult to measure quasi-option value, and there are only a few applications in the environmental economics literature. Fisher and Hanemann (1986) provided an illustrative example of a currently available strain of corn that may have future value. For additional discussions of quasi-option value, see Fisher (1988) and Dixit and Pindyck (1993).

There is also a slightly more elusive set of values that we for sake of simplicity might call symbolic values. The giant "mammuthtrees" of the western U.S., discovered in the 1850s to be older than the Coliseum of Rome, provide an interesting example. These forests have been said to manifest the American republic and its democracy. Historians have also identified the important role of forests as a symbol of liberty for certain Germanic tribes during the Roman empire. From an economic point of view, those values are non-use values that accrue to the relevant members of the population. Other ethics than those based on the anthropocentric view prescribe that natural systems have an intrinsic value that exists quite independent of humans. There is a value of preserving a forest no matter what our preferences as humans might be. The Land Ethic of Leopold extends "the class of morally considerable beings—to systems and not just individuals" (Pearce & Turner, 1990: 232)). According to this ethic, individuals can only possess relative intrinsic values, which are determined by their "relative contribution to biosphere integrity," according to the review by Pearce and Turner (1990). "Ecological economists" have provided the notion of "primary values of ecosystems," which is related, because it appears to be independent of human preferences; see Gren et al. (1994). They argued that one should add values (related to "ecosystem function") over and above those that are due to utility changes.

A framework for valuing forests

Already in 1849, the German Martin Faustmann established what is now known as the Faustmann rule (or Faustmann-Pressler-Ohlin-Samuelsson rule); see Faustmann (1849, transl. 1995). It tells us when to cut a forest to maximize the social value of it, if there are no externalities. This basic result has been generalized to include forest amenities (see Hartman, 1976, for the original result and Johansson & Löfgren, 1985, and Hanley et al., 1997: chap. 11, for surveys). A typical result in this literature is that the socially optimal rotation age is longer compared to the privately optimal rotation age, if there are positive externalities associated with the stock of forest (such as biodiversity or sequestration benefits). A referee of this paper pointed out that the shape of the function describing "extra-market" values is cru-

cial, as shown by Hartman (1976). It may be optimal to cut a forest earlier than expected, because the trees deplete valuable water supplies (this effect being larger, the larger the trees).

It will be useful here to broaden our discussion, so that the theory involves more than finding the optimal rotation age. Following Dasgupta et al. (1995), there are two ways of assessing changes in (aggregate) well-being. One would be to measure the value of changes in the constituents of well-being (utility and freedoms), and the other would be to measure the value of the alterations in the commodity determinants of well-being (goods and services that are inputs in the production of well-being). The former procedure measures the value of alterations in various "outputs," and the latter evaluates the aggregate value of changes in the "inputs" of the production of well-being. A key theorem in modern resource allocation theory is that, provided certain technical restrictions are met, for any conception of aggregate well-being, and for any set of technological, transaction, information, and ecological constraints, there exists a set of shadow (or accounting) prices of goods and services that can be used to obtain an index of welfare change. The index in question has the following property: small investment projects that improve the index are at once those that increase aggregate well-being. Provided the set of accounting prices is unaffected, an improvement in the index owing to an alteration in economic activities reflects an increase in aggregate well-being.

To formalize the above in a simple model, let S be an index of the stock of forests and denote the rate of harvest by X. Assume that the dynamics of the forest can be represented by the growth function q(S,Y), where Y can be interpreted as expenditures on fertilizer or silviculture measures in general (for a more detailed model, see Vincent & Hartwick (1997)). We define the net change of the forest stock as:

$$dS/dt = q(S, Y) - X$$

The production in society is described by the production function

$$W = F(K, S, X),$$

where K is the stock of real capital. We exclude labor in this particular study as it would not give any further insights. Total output (W) can be used for consumption (C), investment in real capital (I), and investments in forest resources (Y):

$$W = C + I + Y.$$

The utility of the representative individual is U(C, S), where U is the utility function. We neglect the flow of environmental services from forests and include only the flow of consumption benefits and forest stock benefits. I will comment on this limitation below. It is assumed that society is behaving as if it is maximizing the present value of future utilities, discounted by $d>0$, subject to technological and ecological constraints:

$$\text{Max} \int U(C, S)e^{-dt}dt.$$

subject to

$$dK/dt = F(K, S, X) - C$$
$$dS/dt = q(S, Y) - X$$

It is a straightforward exercise; see Dasgupta et al. (1995) to obtain an index of welfare from this problem. The index will be called NNP and can be written as (consumption is used as the numeraire):

$$NNP = C + p1*dK/dt + p2*dS/dt + p3*S$$

Where p1, p2, and p3 are the shadow prices on the stocks. The index includes the value of final consumption, the value of the net change in the stock, and a separate value of the stock. This last value occurs because we have assumed that the individual has preference for preservation of the stock of forests, which might be interpreted as a biodiversity value, or an amenity value of virgin forests. As stated above, flow benefits of forest use have been disregarded. Such benefits occur if the individual places a value on the change of the stock per se. Including such values in the model will only affect the shadow prices; the welfare index will have the same structure.

The welfare index includes the current and the future, in the sense that consumption provides well-being "now," and the value of the net change of the stock signals future consumption possibilities. Thus, if we want to measure the "true" contribution to human well-being of forests, we must include additional values over and above those signaled by the market. The construction above is particularly amendable to an extension of the conventional national accounts. By suitable modifications we can use the framework to establish a "green" accounting system at the firm level or to evaluate the social profitability of forest management at the project level (e.g., in a cost-benefit analysis of whether or not a particular forest area should be cut).

Suppose, for example, that we want to use this framework to account for possible acid rain damages in an expanded set of the national accounts (incidentally, the most recent assessment of forest damages of acid rain suggest that earlier concerns may be invalid; see Binkley & Högberg, 1997). It would seem natural to simply deduct the value of the damages caused by acid rain, by calculating the reduction of the growth and the impact on the quality of the trees. Such a calculation has been carried out within the Swedish program on resource accounting; see KI (1998). We could simply deduct the value of such damages from the welfare index to obtain a better picture of the economic and ecological systems, although some care is required due to the risk of double counting. In a market economy such impacts may already be discounted in the market prices of the forest stock. If it is known that all forests in a certain area will be destroyed, we have every reason to believe that this information will be reflected in the market. Consequently, our

standard definition of national product will include this information, at least as long as profits are affected by the quality of the forests. Whether or not the market puts a "correct value" on these damages may be debated, but there is certainly a risk for double-counting the damages, should we simply deduct them in order to obtain a "green" national product.

In a more general model, such as the one provided by Vincent and Hartwick (1997), we could include alternative land uses. Their model highlights the proper valuation of "ecological" services that one sector "delivers" to another. For example, the market value of agricultural products depends on the ecological services provided by forests. The usefulness of the general approach is that it brings out these kinds of issues and allows us to properly estimate the contribution of forests to human well-being, without double-counting.

Empirical Approaches

Techniques to value non-market goods have been rapidly developed over the past two decades. The portfolio of useful approaches has been expanded, and the strengths and weaknesses of each valuation method are now better understood, after more than 30 years of applying them. In what follows, I give a brief account of which methods are useful in the context of valuing forests.

The so-called direct methods of valuation focus on the individual's subjective valuation of a certain resource change, using carefully constructed questionnaires or interviews. The contingent valuation method (CVM) is the most popular among these. At first blush the CVM is straightforward; simply ask a set of people how much they would be willing to pay (WTP) for obtaining a particular good. An application of the CVM needs input not only from economic theory, but also from several other disciplines, including sociology, psychology, statistics, and survey research. Several thousands of applications exist, many of them related to forestry. For a recent survey of the contingent valuation method, see Bateman and Willis (1998) and the special issue of *Journal of Economic Perspectives* (1994).

The CVM and related direct methods, such as conjoint analysis, are currently the only techniques that allow us to estimate non-use values. At the same time, these techniques are subject to the basic weakness that the respondent is answering a hypothetical question. If we abstain from including non-use values, there is some risk for underestimating the "true" values. If we do include them via the CVM, there is a certain risk that the estimate will be biased in some direction.

In certain cases, one can use observed market behavior to infer willingness to pay for environmental quality. The travel-cost method is based on the assumption that the price of enjoying the services of a site depends on the distance to the site, time costs, and other costs related to the visit. The individual will indirectly reveal his willingness to pay for the services of the site by paying the travel costs. By analyzing data on the proportion of visitors from a given number of points (these points are typically defined by the counties surrounding the sites), one obtains a

picture of the demand for the site. The approach has been substantially generalized, since it was first proposed by Harold Hotelling in 1947. He suggested the approach on a direct question from the head of the National Park Service. The original "zoning" approach, where data is based on characteristics of a zone (such as a county), has been replaced by approaches that use data on individuals.

By construction, the travel-cost method can only shed light on values related to expenditures on marketed goods. It is unable to capture passive use values. The practical problems are many, and I can only mention a few of them here. First, it is not clear how the time costs of a trip should be approximated (a priori it is even hard to tell whether the time costs are negative or positive, because the traveler might enjoy the scenery on the way to the site). Furthermore, the trip may have multiple objectives, so that there is a joint-cost problem. It is also difficult to define and incorporate substitute sites into the empirical analysis. I refer to Smith (1996: chaps. 4–15) for extensive discussion of the travel-cost method and to Wibe (1994) for applications related to recreational benefits of forests.

The hedonic pricing method focuses on property values and how amenity benefits may be mapped into those. Two neighboring houses that are identical in every respect, except that one provides a scenic view (such as the Golden Gate), typically fetch different market prices. The hedonic method can be used to distill the values people have been observed to place on amenities. For example, Garrod and Willis (1992) found that different types of forests affect neighboring housing prices. Tyrväinen (1999) examined the relation between property values and urban forestry benefits in Joensuu (a town in eastern Finland with about 49,000 inhabitants). She measured those benefits by using three variables: (i) distance to the nearest wooded recreational area, (ii) direct distance to the nearest "forest park," and (iii) the relative amount of forested areas in the housing district. "Apartment price" is then regressed against these variables and a set of other variables describing the apartment and the location. Tyrväinen (1999: 328) found that a 100-m increase in distance to "wooded recreational areas" reduces "apartment price" by FIM 42 (about USD 6) per meter^2.

An advantage of the hedonic price method is that it focuses on observed behavior and can give estimates of use values. However, the method is marred by several theoretical and practical problems. First of all, it is important that the market under scrutiny is working well, such that the market prices give the correct signals. Second, it has to be assumed that the agents on the housing market have perfect information about the extra-market good. Third, it may be difficult to identify the demand and supply curve for housing. One often assumes that the supply of housing is fixed, which simplifies the analysis considerably. If, however, supply is price-responsive, the method becomes rather more difficult to use in practice.

The avoidance-cost method for approximating environmental damages has been used extensively. The idea is to calculate the cost of restoring the environment to a predefined quality level. Examples include: the cost of using less hazardous chemical products, the costs of purifying emissions, and the costs for liming a lake. Greig and Devonshire (1981) studied the relationship between tree cover and salin-

ity and that between salinity and household expenditure in Kerang, Australia. If water becomes more saline, households would need to spend some resources to keep water quality constant (such as the use of more expensive detergents, installing new water filters, and expenditures on renewing water-using equipment more frequently). If woodland is retained, these expenditures would be unnecessary. Tree cover retention maintains the transpiration and evaporation of water from the soil and therefore minimizes salinization of the water that ends up in the urban water supply. Greig and Devonshire (1981) estimated the amount of money the households would have to spend to keep water quality constant, arguing that this would be a measure of the benefits of retaining tree cover to avoid the increase in salinity of the water.

The weakness of the avoidance-cost method is that it does not necessarily measure what economists are ultimately interested in, i.e., welfare changes. Suppose we use the avoidance-cost method to estimate the noise damage due to an airport construction. We could estimate the cost of installing particular windows that would reduce indoor noise. Clearly, such costs do not include the welfare loss associated with a change of the noise level outdoors. In addition, people may well prefer to have the window open. The essential weakness of any method that focuses on the cost side is that costs do not necessarily measure the total welfare loss. The main advantage of the avoidance-cost method is that it is often inexpensive to use.

An approach here called "the politically determined willingness to pay" presumes that there is a politically determined environmental goal for a particular environmental problem. The annual costs of achieving the goal can be used as an approximation of what the public would be willing to pay to reach it. Unfortunately, explicitly formulated environmental goals do not always exist. If they do exist, we often observe that they change over time. When aspiration levels change dramatically upward, the index of well-being will be reduced, while in the short run environmental quality may not change at all. This makes a time series all the more difficult to interpret. A peculiar artifact of this approach is that the conventional net national product and its "green" counterpart will be equal in a country with no environmental policy; if there are no goals, there can be no cost of reaching them.

An often used approach in the valuation literature is based on opportunity costs. For example, given an estimate of the area of forests that need to be set aside to sustain a given level of biodiversity, the opportunity costs of the "lost" lumber can easily be calculated; see Hultkrantz (1991) for an example. This method does not necessarily provide a good estimate of the underlying welfare change. Suppose that we did use the opportunity costs of "lost" lumber as a value of biodiversity. There is no guarantee that the current market price of timber times the area saved gives a good estimate of the value of preserving biodiversity.

Benefit transfers is a simple approach of extrapolating the values from one study to another, without the need for carrying out a new valuation study. By regression techniques one can extrapolate the results from one study in a particular area to another area. For example, one can study the value people place on preserving old-growth forests in Norway and extrapolate these values to a similar area in

Sweden using regression techniques. This is a "quick-and-dirty" technique that is fraught with statistical and other problems. Its main attractiveness is that it provides an inexpensive way of obtaining rough estimates of environmental benefits.

Applications

In this section, I discuss a number of empirical studies to illustrate some points made earlier in this paper. I organize the studies according to the level of aggregation, beginning with the "world," proceeding to the country level and then, finally, I discuss some studies at the household level. Useful summaries of the relevant literature may be found in Forestry Commission (1999), Vincent and Hartwick (1997), Sharma (1992), and Wibe (1994).

The global value of forest resources

In a much-noted paper in *Nature*, Costanza et al. (1997) estimated the total value of the world's ecosystem services. They identified 17 ecosystem services, ranging from "regulation of atmospheric chemical composition" to "cultural" services. As far as possible, a value was estimated for all categories for a number of biomes, including tropical and temperate/boreal forests. Some of the currently available valuation studies were used to obtain an average per hectare value. According to Costanza et al., the total yearly value of ecosystem services from forests is about USD 969 per ha, which, according to their calculations, would suggest that forest ecosystems contribute about 12% of the total value of the world's ecosystem. Recreational values are found to be on the average USD 86 per ha, and the genetic resources (medicine, etc.) are valued at USD 16 per ha. In total, the value of the world's ecosystem services are claimed to be roughly twice the world's national product.

As pointed out earlier, the information contained in such a calculation is difficult to interpret. Clearly, it would be a gross and costly mistake to use some of these numbers in any decision relating to the husbandry of our environmental and natural resources, for reasons I now explore.

The authors claimed that "it can be instructive to estimate 'marginal' or 'incremental' value of ecosystem services" (Costanza et al., 1997: 263). The authors noted that the total value of all ecological systems must be infinite. Nevertheless, their concept of the total "marginal" value of ecosystem services has little economic meaning, even if restricted to forests. At best, the numbers could be interpreted to measure the utility difference (converted into money) between having and not having the world's forests available for one year. Surely, this number must be very large, and perhaps infinite, considering the fact that "our very biosphere is dependent on the metabolism, death, and recycling of plants, especially trees" (Encyclopedia Britannica, 1999). It follows from this discussion that dividing the estimated value of forest ecosystems by the total number of hectares cannot be interpreted as some kind of average ecosystem value of forests. Hence, I would argue against using these estimates in a cost-benefit analysis.

The national level

Nordhaus and Tobin (1972) proposed adjusting the national product for the disamenities of urban life and other matters, such as the defense system. A number of papers appeared in the 1970s (e.g., Weitzman, 1976; Hartwick, 1977) that dealt with the proper measurement of a sustainable income. Mäler (1991) provided an influential theoretical paper, in which he set out the structure of a "green" accounting system. Mäler (1991) defined the "green" national product rigorously and showed why a correctly computed national product can be given a welfare interpretation. For a recent comprehensive assessment, see Nordhaus and Kokkelenberg (1999).

Perhaps the most influential empirical paper on resource accounting is the WRI study of Indonesia (Repetto et al., 1989), which suggested that the national product is very different if natural degradation is taken into account. The proposed national product calculated by Repetto et al. (1989) is difficult to interpret, mainly because it was not based on a rigorous theoretical model. It excluded a number of important items, such as pollution and depreciation of real capital. There is also an issue of how the authors chose to calculate depreciation of forests. Despite its shortcomings, the study has been an important factor behind the rapid development of the empirical literature in this area, because it showed that resource accounting in monetary terms was possible.

The interest in "green" forest accounts is growing. Vincent and Hartwick (1997) surveyed applications of greening the forest accounts in over 100 countries (the level of aspiration varied considerably in the individual studies). They surveyed country studies from Asia (8 studies), Central and South America (5 studies), Europe (5 studies), North America (4 studies), Oceania (4 studies), and Africa (2 studies). I refer to their survey for details about individual studies and will concentrate on a Swedish study, since it is one of the most ambitious forest-accounting studies that have been carried out this far.

Hultkrantz (1991) made an attempt to adjust the sectoral forest accounts in Sweden by including several non-priced services provided by forests. To the extent possible, Hultkrantz (1991) utilized market prices to evaluate each component. Market data are available for timber, berries, and mushrooms. The value of meat from hunting and recreational values are obtained from a contingent valuation study. Biodiversity is valued by considering the area of protected land that must be set aside to protect biological diversity. Hydrological effects, e.g., forest absorbation of water that could have been used for power generation, are not valued explicitly. Carbon fixing is valued by using the effluent fee of carbon dioxide. Note that Hultkrantz (1991) counted the increase of growing forest stock twice. First, the timber value and then the value of carbon fixing. The annual depletion of exchangeable cations in forest soils can be compensated by liming, and this cost is used as a proxy. For lack of data, nitrogen leaking is not considered explicitly. Forests (in the north of Sweden) also provide reindeer forage. Because changes in lichen stocks are not included in the current accounts, Hultkrantz (1991) utilized studies on opportunity costs to obtain a value of the change is stocks. Note that

the consumption of reindeer forage is already included in the national accounts, because the availability of reindeer forage will affect profits in this industry.

The following table shows an updated version of the Hultkrantz (1991) study carried out by Eliasson (1995), as well as the original numbers. In the same table, forest income from the Finnish forests are also shown (Kriström, 1999: 407).

The most important item in the above list is the carbon dioxide assimilation service. The Intergovernmental Panel on Climate Change lists the value of avoiding a ton of carbon in the atmosphere in the interval $5–$125 and the used value of the Swedish carbon tax is in the middle of that interval.

An important omission from the above calculations is recreational values, and there is every reason to believe that this is a significant omission. Preliminary investigations by the Swedish Bureau of Economic Research suggest that this value may be around USD 2.5 billion. According to Nordhaus and Kokkelenberg (1999: 135), the USDA has estimated that of the total USD 9 billion value of forests goods and services in 1993, production of minerals and timber and grazing range services accounted for about 20%, while recreational and wildlife services provided 80%. These figures also suggest that recreational values from forests may be important. There are also other benefits that are not included in the Swedish study, but which could possibly be included, e.g., the ecosystem functions estimated in Molina and Montoya (1996) (see also the survey by Pearce, 1999: 10).

There are also a number of cost-benefit analyses involving forests at large scales. For example, Van Kooten (1999) analyzed the costs and benefits of reducing the allowable annual cut (AAC) in British Columbia. He found that the cost of reducing the AAC by 10 or 20% is in the range of USD 500–700 million per year. Recreation benefits and other use values are found to be significantly smaller, or up to

Table 1. Modified forest accounts for income from Finnish (1991) and Swedish Forests (1987 and 1991). Billion SEK (1 USD ≈ 8.30 SEK). Source: Finnish Data: Adapted from Nordic Council report (Anon (table 2.1.11, p.31)). Swedish data: Hultkrantz (1991), Eliasson (1995). Sums do not match due to rounding errors.

Item	Finland 1991	Sweden 1987	Sweden 1991
Market value of Timber	18.0	25.3	21.3
Inputs from other sectors	-2.6	-4.3	-4.3
Stock growth	3.3	5.6	5.7
Silviculture	-2.4	-2.1	-2.0
Subtotal 1	16.3	24.5	20.7
Berries	0.1	0.7	0.4
Edible fungi	0.01	0.3	0.3
Lichen	0.4	0.6	0.7
Meat from game	—	0.7	0.8
Subtotal 2	0.9	2.3	2.2
Biodiversity*	-0.6	-1.7	-1.5
Carbon dioxide assimilation	5.3	3.0	2.8
Loss of exchangable cations in soil	—	-1.0	-1.0
Loss of lichen stocks	—	-.03	-0.06
Subtotal 3	4.7	0.4	0.3
TOTAL NET INCOME	21.7	27.3	23.2

nearly USD 13 million per year. While the estimates of non-use values are very uncertain, Van Kooten found that even the most optimistic estimate of the non-market benefits of forests gives a net loss of nearly USD 300 million for reducing AAC to 10 or 20%. A similar cost-benefit analysis was presented in Kriström (1990), focusing on the preservation of certain old-growth forest ecosystems in Sweden. His analysis did not give support to a cutting alternative, even though the total benefits of preservation are relatively small in comparison to similar estimates in the literature.

Contribution of forests at the household level

There are now a large number of studies of non-market forest benefits that arise at the level of the household. I shall only give a glimpse of this activity in this paper. A significant number of studies focus on the recreational value of forests. Wibe's (1994) survey of non-wood benefits in forest included about 200 studies on recreational values. Most studies are based on CVM or the travel cost method. He presented values in the range of $25–$50 per day of recreational visit, where the higher value refers to hunting and the lower to other kinds of recreational activities like camping, hiking, etc.

Hanley and Ruffel (1993) studied the value of UK public forests, in particular the value people attach to different characteristics of the forests. They found that WTP varies with characteristics such as height diversity, the percentage of broad-leaved trees, and the percentage of forests as open space. A similar study has been undertaken by Li and Mattsson (1994). They interviewed a set of Swedish respondents about their WTP for different types of forests. The respondents were specifically asked to rank different regeneration programs. The Swedish forest landscape is dominated by Scotch pine and Norwegian spruce (86%). Respondents preferred a more varied landscape, with a higher share of broad-leaved trees.

There are a couple of studies that shed light on non-use values of forests. In a study by Hagen et al. (1992), it was found that the value of preserving an old-growth ecosystem (the value of the spotted owl habitat) could be as high as USD 200 per household. Van Kooten (1999) used a value of USD 300 per household per year, which is "high compared to other studies" (Van Kooten, 1999: 72). Adamowicz et al. (1998) estimated non-use values of protecting old-growth forests (via the protection of the threatened caribou). In my study of preserving (certain) old-growth ecosystems in Sweden (Kriström, 1990), I found that the total willingness to pay is in the order of USD 10–20 per household per year. Willingness to pay was calculated for four different (non-exclusive) preservation motives. Those respondents who were willing to pay for preserving the areas, but claimed no interest in visiting them, have lower willingness to pay than households that expressed only "personal use motives" for preservation. The highest willingness to pay is found in the group that express both "user" and "non-user" motives for preservation. This response pattern is intuitively plausible, but cannot be used to suggest any relation between use and non-use values, in terms of their share of total value. The reason

is technical and will not be pursued here. The interested reader is referred to Johansson (1993) for details about the path-dependency problem involved.

Is Valuation Useful?

Valuation is useful if it helps society to make better decisions about the proper husbandry of our environments and natural resources. Carefully structured valuation studies based on received economic theory could be useful in this endeavor. Valuation studies that lack theoretical underpinnings, or are disjunct from an underlying resource allocation problem, may, however, do more harm than good. This is an opinion of which I can offer no proof. I simply believe that measurement without theory is as dangerous in the area of valuation as it may be elsewhere in the social and other sciences. I strongly believe that valuation exercises should not be viewed as being independent of underlying resource allocation problems. It may be of interest to know that preserving a forest has a value, but only so far as we are able to compare this use of a forest to another use. In the market for private goods, we continually make comparisons and decide that some allocation of spending is better than another. Valuation of non-market goods should be seen in the same light, and not as a vehicle for providing values unrelated to choice. For this reason, I find valuation studies that purport to measure the global value of forests of limited interest. In addition, we have no sound theoretical basis for averaging such global values and use them at a smaller scale.

I believe that a comprehensive assessment of how forests contribute to human well-being will often be a useful input to reasoned discussion involving the proper use of our forest resources. Even so, it is important to realize that economic analysis in this context is not reduced to finding the "true" value of a certain project, summarized in a single number. Many of the approaches discussed in this paper allow us to shed light on "who wins" and "who loses" from a certain change of resource use. This kind of information, coupled with other ways of shedding light on resource allocation problems, allows us to make better informed decision. The bottom line is that valuation is and always will be an input to a decision process.

Acknowledgments

Invited paper, "Managing Human-dominated Ecosystems," Missouri Botanical Garden, March 26–28, 1998. The paper has been much improved by the insightful comments of two anonymous reviewers.

Literature Cited

Adamowicz, W., P. Boxall, M. Williams & J. Louviere. 1998. Stated preference approaches for measuring passive use values: Choice experiments and contingent valuation. American Journal of Agricultural Economics 80(1): 64–75.

Bateman, I. J. & K. G. Willis (Editors). 1998. Contingent Valuation of Environmental Preferences: Theory and Practice in the USA, Europe, and Developing Countries. Oxford Univ. Press, Oxford.

Binkley, D. & P. Högberg. 1997. Does atmospheric deposition of nitrogen threaten Swedish forests? Forest Ecology and Management 92: 119–152.

Bishop, R. C. 1982. Option value: An exposition and extension. Land Economics 58: 1–15.

Ciriacy-Wantrup, S. V. 1986. *In* R. C: Bishop & S. O. Anderson (editors), Natural Resource Economics: Selected Papers. Westview Press, Colorado.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. J. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Daily, G. (Editor). 1997. Nature's Services: Society's Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Dasgupta, P., B. Kriström & K.-G. Mäler. 1995. Current issues in resource accounting. *In* P.-O. Johansson, B. Kriström & K.-G. Mäler, Current Issues in Environmental Economics. Manchester Univ. Press, Manchester, U.K.

Dixit, A. K. & R. S. Pindyck. 1993. Investment Under Uncertainty. Princeton Univ. Press, Princeton, New Jersey.

Eliasson, P. 1995. Miljöjusterade Nationalräkenskaper för den Svenska Skogen åren 1987 och 1991. Department of Forest Economics, SLU, Umeå, Sweden.

Economist, The (2/20/1999). Ethnobotany. Shaman loses its magic.

Encyclopedia Brittanica. 1999. "Tree" Encyclopædia Britannica Online. <http://search.eb.com/bol/topic?idxref=93972> [Accessed 16 March 1999].

FAO. 1993. The Challenge of Sustainable Forest Management: What Future for the World's Forests? FAO, Rome.

Faustmann, M. [1849, transl. 1995.] Calculation of the value which forest land and immature stands possess for forestry. Journal of Forest Economics 1(1): 7–44.

Fisher, A. C. 1988. Key aspects of species extinction: Habitat loss and overexploitation. *In* V. K. Smith (editor), Environmental Resources and Applied Welfare Economics: Essays in Honor of John V. Krutilla. Resources for the Future, Washington, D.C.

Fisher, A. C. & W. M. Hanemann. 1986. Option value and the extinction of species. *In* V. K. Smith (editor), Advances in Applied Microeconomics, Vol. 4. JAI Press, Greenwich, California.

Forestry Commission. 1999. The Living Forest: Non-Market Benefits of Forestry. The Stationery Office, London, U.K.

Friedman, M. 1962. Capitalism and Freedom. Univ. Chicago Press, Chicago.

Garrod, G. & K.Willis. 1992. "Valuing Goods" characteristics: An application of the Hedonic Price Method to environmental attributes. Journal of Environmental Management 34: 59–76.

Gren, I.-M., C. Folke, K. Turner & I. Bateman. 1994. Primary and Secondary Values of Wetland Ecosystems. Environmental & Resource Economics 4: 55–74.

Greig, P. J. & P. G. Devonshire. 1981. Tree removals and saline seepage in Victorian catchments—Some hydrologic and economic results. Australian Journal of Agricultural Economics 25(2): 134–148.

Grimes, A., S. Loomis, P. Jahnige, M. Burnham, O. Onthank, R. Alarcon, W. Palacios Cuenca, C. Caron Martinez, D. Neill, M. Balick, B. Bennett & R. Mendelsohn.

1994. Valuing rain forest: The economic value of nontimber forest products in Ecuador. Ambio 23(7): 405–410.

Hagen, D. A., J. W. Vincent & P. G. Welle. 1992. Benefits of preserving old-growth forests and the spotted owl. Contemporary Policy Issues 10: 13–26.

Hanley, N. & R. Ruffell. 1993. The contingent valuation of forest characteristics. *In* W. Adamowitz & W. White (editors), Forestry and the Environment: Economic Perspectives. CAB International, Wallingford, Oxfordshire, U.K.

Hanley, N., J. F. Shogren & B. White. 1997. Environmental Economics in Theory and Practice. MacMillan, London.

Hartman, R. 1976. The harvesting decision when a standing forest has value. Economic Inquiry 14: 52–58.

Hartwick, J. M. 1977. Intergenerational equity and the investing of rents from exhaustible resources. American Economic Review 67(5): 972–974.

Hotelling, H. 1947. The Economics of Public Recreation: The Prewitt Report. National Parks Service, Washington.

Hultkrantz, L. 1991. National accounts of timber and forest environmental resources in Sweden. Environmental and Resource Economics 2: 283–305.

Johansson, P.-O. 1993. Cost Benefit Analysis of Environmental Change. Cambridge Univ. Press, Cambridge.

Johansson, P.-O. & K.-G. Löfgren. 1985. The Economics of Forestry and Natural Resources. Basil Blackwell, Oxford.

Journal of Economic Perspectives, Fall, 1994. Symposium on contingent valuation. Vol. 8(4).

Kanninen, B. J. & B. Kriström. 1992. Welfare Benefit Estimation and the Income Distribution. Beijer Institute of Ecological Economics, Working Paper # 21.

KI. 1998. SWEA, Swedish Economic and Environmental Accounts. National Institute of Economic Research (KI), Stockholm.

Kriström, B. 1990. Valuing environmental benefits using the contingent valuation method: An econometric analysis. Umeå Economic Studies, 219. University of Umeå.

Kriström, B. 1999. On the incorporation of non-market outputs of forests into national accounting systems. *In* S. S. Roper & A. Park (editors), The Living Forest: Non-Market Benefits of Forestry. Forestry Commission, The Stationery Office, London, U.K.

Krutilla, J. A. 1967. Conservation reconsidered. American Economic Review 57: 777–786.

Li, C.-Z. & L. Mattsson. 1994. How do different forestry practices affect the nontimber value of forests. Journal of Environmental Management 41: 79–88.

Mäler, K.-G. 1991. National accounts and environmental resources. Environmental & Resource Economics 1: 1–15.

Milgrom, P. 1992. Is sympathy an economic value? Philosophy, economics and the contingent valuation method. *In* Contingent Valuation: A Critical Assessment. Cambridge Economics, Inc.

Molina, J. P. & C. Montoya. 1996. Valuation of regional natural resources in Colombia. A case study. Mimeo, Beijer Institute of Ecological Economics, Stockholm, Sweden.

National Oceanic and Atmospheric Administration (NOAA). 1993. Report of the NOAA-panel on contingent valuation. Federal Register 58: 10, January 11, 1993. 4602–4614.

Nordhaus, W. D. & J. Tobin. 1972. Is growth obsolete? Volume 5, Proceedings of the Fiftieth Anniversary Colloquium of the National Bureau of Economic Research, New York.

Nordhaus, W. D. & E. C. Kokkelenberg. 1999. Nature's Numbers: Expanding the National Economic Accounts to Include the Environment. National Academy Press, Washington, D.C.

Olembo, R. 1995. A brief overview of forests. *In* M. Atchia & S. Tropp, Environmental Management: Issues and Solutions. Wiley, New York.

Pearce, D. 1999. Can non-market values save the world's forests. *In* S. S. Roper & A. Park (editors), The Living Forest: Non-Market Benefits of Forestry. Forestry Commission, The Stationery Office, London, U.K.

Pearce, D. & R. K. Turner. 1990. Economics of Natural Resources and the Environment. Harvester Wheatshef, New York.

Repetto, R., W. Magrath, M. Wells, C. Beer & F. Rossini. 1989. Wasting Assets: Natural Resources in the National Income Accounts. World Resources Institute, Washington, D.C.

Sharma, N. P. 1992. Managing the World's Forests: Looking for Balance Between Conservation and Development. Kendall/Hunt, Dubuque, Iowa.

Smith, V. K. 1996. Estimating Economic Values for Nature. Edward Elgar, Cheltenham, U.K.

Tyrväinen. 1999. Economic estimates of urban forests: Comparison of economic estimates. *In* Forestry Commission, The Living Forest: Non-Market Benefits of Forestry. The Stationery Office, London, U.K.

Van Kooten, R. C. 1999. Preserving species without an Endangered Species Act: British Columbia's Forest Practices Code. *In* M. Boman, R. Brännlund & B. Kriström (editors), Topics in Environmental Economics. Kluwer Academic Press, Amsterdam.

Vincent, J. & M. Hartwick. 1997. Forest Resources and the National Income Accounts: Concepts and Experience. FAO, Rome.

Weisbrod, B. A. 1964. Collective-consumption services of individual consumption goods. Quarterly Journal of Economics 78: 471–477.

Weitzman, M. L. 1976. On the welfare significance of national product in a dynamic economy. Quarterly Journal of Economics 90(1): 156–162.

Wibe, S. 1994. Non-Wood Benefits in Forestry—Survey of Valuation Studies. Working Paper 199, Department of Forest Economics, SLU-Umeå, Sweden.

The Value of Water and Watersheds: Reconciling Ecological and Economic Externalities

Edward B Barbier
John S Bugas Professor of Natural Resource and
Environmental Economics
Department of Economics and Finance
P.O. Box 3985
University of Wyoming
Laramie, Wyoming 82071-3985, U.S.A.

Abstract

External ecological support functions that appear to originate from one ecosystem but essentially support and protect the functioning and productivity of another system can be referred to as ecological externalities. Economists are particularly interested in the class of ecological externalities that also contribute to important economic values, albeit indirectly, through the support and protection of economic activity and property. This paper is concerned with three examples of ecological and economic externalities commonly found in watershed and river basin systems: the diversion of water by upstream water projects, increased runoff and sedimentation caused by erosion from upland farming, and improved water yields from upper catchment timber management regimes. Simple analytical models are developed to highlight the key features of each type of watershed externality. Case study examples are also explored to illustrate the main valuation and incentive issues associated with each externality: upstream water diversion in the Hadejia-Jama'are River Basin, northern Nigeria; upper watershed degradation on farmlands in the uplands of Java, Indonesia; and joint production of timber and water yields in the upper catchment of the Colorado River Basin, U.S.A. As these case studies demonstrate, valuation of watershed externalities cannot be separated from the need to design effective economic strategies and appropriate incentives to manage these externalities. Hopefully, the general principles derived from the analysis of watershed externalities in this paper are also applicable to other cases where external ecological support functions yield important economic values.

1. Introduction: Ecological and Economic Externalities

Ecologists define ecosystems as biological communities that interact with their non-living or abiotic environment (Beeby 1993). However, ecologists also recognize that ecosystems are not isolated but frequently interact with each other. In particular, the ecological functioning of one ecosystem may affect the functioning of an adjacent system. As a result, the overall productivity and stability of an ecosystem may be critically dependent on the maintenance of a few key external support functions provided by a neighboring ecosystem. Examples include the role of coastal marshland and mangrove systems as breeding grounds and nurseries for offshore fisheries; flood and sedimentation control provided by upper watershed and montane forest systems; sediment and nutrient retention by riverine wetlands; and semiarid and arid brush forests protecting against desertification of rangelands. In this paper, those external ecological support functions that appear to originate from one ecosystem but essentially support and protect the functioning and productivity of another system will be referred to as "ecological externalities."

Of particular interest to economists is the class of ecological externalities that also contribute to important economic values, albeit somewhat indirectly, through supporting and protecting ecosystems that are in turn exploited through human economic activity. For instance, offshore fisheries supported by mangrove breeding habitats may be harvested commercially, recreationally, and for subsistence consumption; flood and sedimentation control by upland forests may protect lowland residential areas, industries and agricultural production; the waste treatment function of wetlands may reduce human health risks; and rangelands protected from desertification may provide grazing for livestock herds. Increasingly, economists are referring to such external ecological support and protection functions as good examples of indirect use value, or the value of the environment as a productive input (Aylward & Barbier, 1992; Barbier, 1994; Freeman, 1993). That is, these ecological functions may have an economic value that arises through their support of economic production and human welfare, or through protection of valuable assets and property. However, as this contribution is non-marketed, goes financially unrewarded, and is only indirectly related to economic activities, the indirect use values of external ecological support functions are often extremely difficult to value.

Nevertheless, it is often important to assess the indirect use value of many external ecological support functions because:

- the lack of markets for these functions means that those individuals benefiting from economic activities supported by the ecological "supply" of these functions have no automatic influence over the quantity and quality of this supply, even though this supply affects their well being
- the supply of these functions are in fact determined by the resource allocation decisions of another set of individuals, whose activities impact on the ecosystems where these ecological support functions originate, but who have no in-

centive to take into account the effects of these ecological externalities on the well being of individuals elsewhere.

For example, fish farmers who convert mangroves to shrimp ponds do not take into account the economic implications of the loss of breeding or nursery habitat for commercial fisheries; loggers contributing to upland deforestation do not incur the costs of dredging downstream reservoirs of any resulting increase in sedimentation; and those who cut brush and trees for fuelwood do not necessarily face the economic consequences of increased wind erosion on grazing rangelands. Such situations describe the classic "market failure" symptom that economists refer to as an *economic externality* (Cornes & Sandler, 1986). Thus, under certain conditions, the existence of ecological externalities also gives rise to economic externalities.

This paper is concerned with a particular setting in which ecological and economic externalities might simultaneously occur—the flow of water and sediment through the water catchment and drainage basin area of a river system—and the influence of certain economic activities on these ecological functions. The next section describes briefly the unique ecological characteristics of river systems and their watersheds, and identifies three different types of ecological and economic externalities that influence the delivery of water and sediment in this system: the diversion of water from downstream activities through upstream irrigation and water development projects; increased sedimentation of rivers through upland farming practices that cause soil erosion; and management of upper watershed forest plantations for both timber production and increased water yields. The first case is a straightforward example of a negative ecological and economic externality. The second case is also an example of a negative externality, but as upland farmers may also experience "on-site" costs of soil erosion, they may have at least a partial incentive to control some of the soil erosion. The final case is actually an example of a positive ecological and economic externality.

As the nature of the ecological and economic externality differs for all three cases, each offers a unique insight to the particular type of *valuation* issues involved, and more importantly, the key *incentives* problem that need to be addressed if the flow of water or sediments through the entire river system are maintained so as to ensure maximum net economic benefits to all of those who depend on the catchment and drainage basin system. To highlight these valuation and incentive issues, the paper will illustrate each type of ecological and economic externality with a corresponding case study example. Hopefully, these brief examples will also provide some useful guidance for the analysis of ecological and economic externalities that occur between other types of ecosystems and economic activities.

2. Watersheds, Water, and Externalities

The various river and lake ecosystems found along the course of a watershed catchment and drainage basin area must continually respond and adjust to the water,

sediment, and nutrient flows through the catchment (Wilcock, 1993). Under natural conditions and over long periods of time, the productivity, plant communities, and food chains that form the various ecosystems located in the catchment must adapt to the physical and chemical conditions, which are in turn influenced by streamflow, sediment transport rates, and nutrient supplies. The upper and lower catchments are dominated by characteristic and often distinct flow conditions.

Although it is possible to view the entire watershed catchment and drainage basin area as a single, "mega" system, ecosystems found in the lower watershed are likely to have different characteristics compared to ecosystems in the upper watershed. For example, in upland catchments, river gradients tend to be steep, streamflow is turbulent and well oxygenated, and sediment load is coarse. In contrast, lowland rivers and waterways will generally have gentle gradients, greater depth, more but finer sediment, and greater biological productivity. Most importantly, lower catchment rivers and ecosystems receive their water discharge and nearly all their sediment from the upper watershed, and over long periods of time, adjust and adapt to these flows of water and sediment from the upper catchment (Wilcock, 1993). In other words, the maintenance of a characteristic downstream transfer of sediment and water is an important external ecological support function that the upper catchment provides for the lower catchment ecosystems.

Needless to say, economic activities in upland areas can affect this external support function by influencing transfer of water and sediment downstream. This will in turn impact on the various economic activities in the lower watershed that may be dependent on lowland ecosystems and water supplies. For example, diversion of upstream water supplies for irrigation, water supply, flood control, and hydroelectricity generation will interrupt the downstream transfer of water and sediment which would otherwise take place, and in particular will affect the characteristic variability of flow to which instream and riparian organisms have adapted. The result is a change in channel flow regime and morphology, sediment transport rates, water quality, and water temperatures, which will have dramatic consequences to the ecology of lowland river ecosystems and floodplains. If the latter disruptions have a negative impact on downstream economic activities, such as irrigated agriculture, fishing, and recreational activities, then there could be significant costs involved. Similarly, road building, deforestation, and farming activities that increase the amount of soil erosion in upland areas could also lead to substantial increases in sedimentation loads and transfers downstream. As a result, in the lower catchment there may be additional economic damages to dams and reservoirs, irrigation networks, and fish stocks. However, not all upland activities that influence the transfer of water and sedimentation downstream may necessarily cause negative economic impacts. For example, in certain upper watershed catchment areas it might be possible to design timber plantation and harvesting systems so that the timing of the harvest will augment water releases and yields. Provided that the dangers of flooding are minimal, the augmentation of catchment water releases may be beneficial in river basins where much of the existing water flow is already being utilized, and if there are problems with chronic seasonal shortages.

The above three examples demonstrate that either augmenting or disrupting the beneficial "ecological externalities" associated with downstream transfer of water and sediment can also result in significant economic externalities. Nevertheless, each type of watershed externality poses very different economic valuation and incentive challenges. The rest of this paper will highlight these challenges. First, however, the key features of each type of watershed externality are explored analytically.

For simplicity, assume that the lower watershed ecosystem is an agricultural system, or an "agroecosystem," dependent on the stock of water available downstream, e.g., stored in a naturally occurring floodplain or in a human-made reservoir serving an irrigation network. This downstream water supply, W, is freely available to the agricultural system, which uses a fixed proportion of it, kW. Output per hectare in the agroecosystem, h_1, is therefore a function of the available water stock, W, and a vector of other inputs, z_1 (e.g., labor, purchased inputs, etc.). Assuming that the agricultural system produces for a market in response to a given market price, p^h, and faces a vector of given input prices, c, the discounted economic returns of present and future production in the agroecosystem can be represented as

$$\begin{aligned} V^1 &= V^1(z_1, W, t;\ p^h, c, \delta) = \int_0^T (p^h h_1 - cz_1)\, e^{-\delta t}\, dt \\ h_1 &= h_1(z_1, W),\ h_{1i} > 0,\ h_{1ii} < 0,\ i = z_1, W, \end{aligned} \tag{1}$$

where δ is the rate of discount and T is the time period over which the agroecosystem is in operation.

Based on the discussion of ecological externalities, assume that the stock of water available downstream and over time is dependent on the influence of an important external ecological support function that the upper watershed catchment provides for the lower catcment—the maintenance of a characteristic downstream transfer of sediment and water. Denoting this external watershed support function as ε, then the impact of this function on the stock of water available downstream can be defined as $f(\varepsilon)$. However, also assume that, at least initially in the system, this externality has been maintained at a constant level, ε_0, for some time, and thus the accumulation of water downstream has adjusted naturally to this level of support, i.e., $f(\varepsilon_0) = 0$. Thus the conditions determining changes in the availability of water downstream over time are

$$\begin{aligned} &W(t) - W(0) = \int_0^t f(\varepsilon)\, dt \\ &\text{or}\quad \dot{W} = f(\varepsilon), \\ &\text{and}\quad \dot{W} = f(\varepsilon_0) = 0,\ W(t) = W(0) = W_0,\ \text{for}\ \ \varepsilon = \varepsilon_0 . \end{aligned} \tag{2}$$

As long as the external watershed support function is maintained at its long run level, ε_0, then the stock of downstream water will also be maintained at its long run initial

supply level, $W(0) = W_0$. This in turn means that the downstream agroecosystem will be able to continue using kW_0 amount of this water as an input into production.

However, as the above discussion indicates, the introduction of economic activities in the upper watershed is likely to influence the transfer of water and sediment downstream. We can represent this impact as $\varepsilon(d)$, where d is determined by the type of upland activity affecting the watershed protection function. For instance, corresponding to the three examples of upland activities discussed above, we can envision three different cases:

Case I Upstream water diversion

In this case, the continuous diversion of water for upstream projects, such as irrigation or water supply, reduces the flow of water through the watershed and drainage basin, directly affecting the supply of water available downstream for the lower watershed agroecosystem. Consequently, this situation could be represented by modifying equation (2) in the following manner

$$W(t) - W(0) = \int_0^t f(\varepsilon(d))dt, \quad f(\varepsilon(d)) = -d$$
$$\text{or} \quad \dot{W} = f(\varepsilon(d)) = -d. \tag{3}$$

Thus d represents the amount of water diverted each period by upstream projects, and hence the amount of water no longer available for the stock of downstream supply, W.

Case II Upland soil erosion

In the second case, the source of disruption to the orderly transfer of water and sediment from the upper to the lower watershed is increased soil erosion and sedimentation from an upland activity, such as farming. In this case, increased sedimentation of river beds, waterways, and reservoirs may reduce the supply of water downstream, but the impact may be less direct than the diversion of water. Thus (2) may take the more general form

$$W(t) - W(0) = \int_0^t f(\varepsilon(d))\,dt, \quad f(\varepsilon(d)) = -g(d), \; g_d > 0, \; g_{dd} < 0$$
$$\text{or} \quad \dot{W} = f(\varepsilon(d)) = -g(d). \tag{4}$$

Here, d is the amount of soil runoff from an upland farming system, and the functional form of $g(d)$ will depend on how sedimentation loads throughout the watershed and drainage basin accumulate and affect the downstream supply of water in the irrigation reservoir or floodplain.

Case III Upper catchment timber harvesting

In the final case, timber harvesting in the upper watershed may actually increase the overall water flow through the catchment, thus augmenting the amount of water available downstream. Denoting v as the volume of timber yield per hectare, equation (2) would now be

$$W(t)-W(0)=\int_0^t f(\varepsilon(d))\,dt,\quad f(\varepsilon(d))=d(v),\ d_v>0,\ d_{vv}<0$$
$$\text{or } \dot{W}=f(\varepsilon(d))=d(v). \tag{5}$$

Effectively, the rate at which downstream water supplies are augmented (d) is a function of timber yields in the upper catchment, $d(v)$. Note that if timber harvesting augments water yields in the watershed, it is also likely to lead to problems of soil runoff and sedimentation. This is particularly the case in tropical forestry (Barbier et al., 1994). Curiously, forestry studies that examine the joint management of forestry for both timber and augmenting water yields generally do not consider that increased water yields from harvesting may also be associated with greater downstream sedimentation (Bowes & Krutilla, 1989; Calish et al., 1978). Although to keep the third case simple this latter effect is ignored, it could easily be incorporated through invoking a combination of equations (4) and (5): $dW/dt = d(v) - g(s(v))$, where $s(v)$ represents the amount of soil runoff associated with timber harvesting.

As equations (3) to (5) indicate, the three upper basin activities have very different impacts on the external ecological support function determining the availability of water in the lower catchment for the downstream agroecosystem. These differing impacts are key to understanding the economic valuation and incentive problems that are unique to each of the above cases. The following sections will explore these three examples of watershed externalities further, both theoretically and with reference to case study examples.

3. Upstream Water Diversion

As indicated by equation (3) for Case I, the type of ecological and economic externalities associated with upstream water diversion are fairly straightforward, "unidirectional" problems. Water that is diverted upstream means less water downstream, and the result is that upstream activities benefit at the expense of downstream activities. Determining the optimal level of water diversion between upstream and downstream uses is therefore a critical issue. By incorporating equation (3) in a simple model of the river basin, it is possible to illustrate the key conditions underlying this important allocation decision. A case study of the impacts of upstream water project developments on the downstream floodplain in the Hadejia-Jama'are River Basin in northern Nigeria also illustrates the valuation and incentive issues associated with this particular type of externality.

If it is assumed that water in a river basin is being continuously diverted upstream for an irrigated agricultural system, then the discounted returns per hectare for the latter system, V^2, may take the following form

$$V^2 = V^2(z_2, d, t;\ p^h, c, \delta) = \int_0^T (p^h h_2 - cz_2) e^{-\delta t}\, dt$$
$$h_2 = h_2(z_2, d, \textstyle\int d\, dt) = h_2(z_2, d, W(0) - W(t)),\ h_{2i} > 0,\ h_{2ii} < 0,\ i = z_2, d, W(0) - W(t), \tag{6}$$

where agricultural yield, h_2, is presumed to be an increasing function of both current water diversion, d, cumulative diversion into the upstream irrigation network, $\int d\, dt$, and variable inputs, z_2. Output is sold at the given market price for agricultural crops, p^h, and the cost of inputs is c. In this simple example, it is assumed that there is no user charge imposed on the agricultural system for the irrigation water supplied through the upstream water project.

In this case, the economic externality problem arises because the farmer benefiting from the diverted irrigation from the upstream water project is unconcerned about any resulting impacts on downstream water availability. For example, it is clear from equation (6), that the economic agent would maximize discounted returns by choosing a rate of water diversion, d^i, in each period t, that would satisfy the following condition

$$\frac{\partial V^2}{\partial d} = p^h h_{2d}(d^i) = 0, \quad \forall\, t \tag{7}$$

i.e., from the standpoint of the upstream irrigation farmer, optimal diversion of water to the upstream irrigated agricultural system in every time period should occur until there are no more current economic gains to be had from using additional water inputs.

In contrast, if there was a single river basin planning authority, this agency would be concerned with maximizing not only the returns to upstream irrigated agriculture, e.g., equation (6), but also the returns to the downstream agroecosystem as well, e.g., equations (1) and (3). Denoting λ as the costate variable, or "shadow price," of the downstream water supply, the current value Hamiltonian, H, and relevant first order conditions of concern to the river basin authority might be

$$H = (p^h h_1 - cz_1) + (p^h h_2 - cz_2) - \lambda d$$
$$\frac{\partial H}{\partial d} = p^h h_2 d(d^*) - \lambda = 0$$
$$-\frac{\partial H}{\partial W} = \dot{\lambda} - \delta\lambda = p^h h_{2W(0)-W(t)} - p^h h_{1W} \quad \text{or} \quad \frac{\dot{\lambda}}{\lambda} + \frac{h_{1W}}{h_{2d}} - \frac{h_{2W(0)-W(t)}}{h_{2d}} = \delta.\ \forall t \tag{8}$$

In comparison with (7), the first order conditions in (8) show that the optimal rate of water diversion for the entire river basin occurs where the marginal benefit to

upstream irrigated agriculture of diversion equals the "shadow price," or value, of water supply to the downstream agroecosystem. By definition this shadow price, $\lambda(t)$, is the imputed value of the downstream water stock, W, in terms of the net returns to the agroecosystem in the lower catchment. As the latter value is positive, and is likely to be increasing over time as W is depleted, then the optimal rate of diversion, d^*, for the entire watershed and drainage basin will be less than the rate, d^i, that an upstream farmer would decide on his or her own.

Consequently, the valuation problem facing the river basin planner is to determine the contribution of the available downstream water to the returns to the lower catchment agricultural system, and how these returns might change over time as more water is diverted to the upstream project. The incentives problem is to ensure that upstream water users do not divert more water than the optimal rate, d^*, which means that the value of water in terms of downstream benefits must be incorporated in decisions to divert water for upstream use. As the following example indicates, overcoming this incentive problem and ensuring an optimal rate of diversion in a river basin system is particularly difficult if plans and development for upstream water diversion are already well advanced.

Case Study: Hadejia-Jama'are River Basin, Northern Nigeria

The following case study is based on Barbier and Thompson (1998). See also Barbier et al. (1993).

In Northeast Nigeria, an extensive floodplain has been created where the Hadejia and Jama'are Rivers converge to form the Komadugu Yobe River, which drains into Lake Chad. Although referred to as wetlands, much of the Hadejia-Jama'are floodplain is dry for some or all of the year. Nevertheless, the floodplain provides essential income and nutrition benefits in the form of agriculture, grazing resources, non-timber forest products, fuelwood, and fishing for local populations. The wetlands also serve wider regional economic purposes, such as providing dry-season grazing for semi-nomadic pastoralists, agricultural surpluses for Kano and Borno states, groundwater recharge of the Chad Formation aquifer, and "insurance" resources in times of drought. In addition, the wetlands are a unique migratory habitat for many wildfowl and wader species from Palaearctic regions, and contain a number of forestry reserves.

However, in recent decades the Hadejia-Jama'are floodplain has come under increasing pressure from drought and upstream water developments. The maximum extent of flooding has declined from between 250,000 to 300,000 ha in 1960s and 1970s to around 70,000 to 100,000 ha more recently. Drought is a persistent, stochastic environmental problem facing all sub-Saharan arid and semiarid zones, and the main cause of unexpected reductions in flooding in drought years. The main long-term threat to the floodplain is water diversion through large-scale water projects on the Hadejia and Jama' are Rivers. Upstream developments are affecting incoming water, either through dams altering the timing and size of flood flows or through diverting surface or groundwater for irrigation. These develop-

ments have been taking place without consideration of their impacts on the Hadejia-Jama'are floodplain or any subsequent loss of economic benefits that are currently provided by use of the floodplain.

The largest upstream irrigation scheme at present is the Kano River Irrigation Project (KRIP). Water supplies for the project are provided by Tiga Dam, the biggest dam in the basin, which was completed in 1974. Water is also released from this dam to supply Kano City. The second major irrigation scheme within the river basin, the Hadejia Valley Project (HVP), is under construction. The HVP will be supplied by Challawa Gorge Dam on the Challawa River, upstream of Kano, which was finished in 1992. Challawa Gorge may also provide water for Kano City water supply. A number of small dams and associated irrigation schemes have also been constructed or are planned for minor tributaries of the Hadejia River. In comparison, the Jama'are River is relatively uncontrolled with only one small dam across one of its tributaries. However, plans for a major dam on the Jama'are at Kafin Zaki have been in existence for many years, which would provide water for an irrigated area totaling 84,000 ha. Work on Kafin Zaki Dam has been started and then stopped a number of times, most recently in 1994, and its future is at present unclear.

A combined economic and hydrological analysis was recently conducted to simulate the impacts of these upstream projects on the flood extent that determines the downstream floodplain area (Barbier & Thompson, 1998). The economic gains of the upstream water projects were then compared to the resulting economic losses to downstream agricultural, fuelwood, and fishing benefits.

Table 1 indicates the scenarios that comprise the simulation. Since Scenarios 1 and 1a reflect the conditions without any of the large-scale water resource schemes in place within the river basin they are employed as baseline conditions against which Scenarios 2–6 are compared. Scenario 2 investigates the impacts of extending the KRIP-I to its planned full extent of 22,000 ha without any downstream releases. In contrast, Scenario 3 simulates the impacts of limiting irrigation on this project to the existing 14,000 ha to allow a regulated flood from Tiga Dam in August to sustain inundation within the downstream Hadejia-Jama'are floodplain. Challawa Gorge is added in Scenario 4 and the simulated operating regime involves the year-round release of water for the downstream HVP, but not for sustaining the Hadejia-Jama'are floodplain. Scenario 5 simulates the full development of the four water resource schemes without any releases for the downstream floodplain. In direct comparison, Scenario 6 shows full upstream development, but less upstream irrigation occurs in order to allow regulated water releases from the dams to sustain inundation of the downstream floodplain.

In Table 2.a., the impacts of Scenarios 2–6 upon peak flood extent downstream are evaluated as the difference between maximum inundation predicted under each of these scenarios and the peak flood extents of the two baseline scenarios (see Barbier & Thompson, 1998, for further details). The gains in upstream irrigated area are also indicated for each scenario in Table 2.a. The estimated floodplain losses are indicated in Table 2.b. for each scenario compared to the baseline Sce-

Table 1. Scenario for upstream projects in the Hadejia-Jama=are River Basin, northern Nigeria.

Scenario (Time Period)	Dams	Regulated Releases ($10^6 m^3$)	Irrigation Schemes
1 (1974-1985)	Tiga	Naturalised Wudil flow (1974-1985)	No KRIP-I
1a (1974-1990)	Tiga	Naturalised Wudil flow (1974-1990)	No KRIP-I
2 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
3 (1964-1985)	Tiga	400 in August for sustaining floodplain	KRIP-I at 14 000 ha
4 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
	Challawa Gorge	348 yr^{-1} for HVP	
	Small dams on Hadejia tributaries		
5 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
	Challawa Gorge	348 yr^{-1} for HVP	
	Small dams on Hadejia tributaries		
	Kafin Zaki	None	84 000 ha
	HVP	None	12 500 ha
6 (1964-1985)	Tiga	350 in August	KRIP-I at 14 000 ha
	Challawa Gorge	248 yr^{-1} and 100 in July	
	Small dams on Hadejia tributaries		
	Kafin Zaki	100 per month: Oct-Mar and 550	
	HVP	in August	None
		Barrage open in August	8 000 ha

Notes: KRIP-I = Kano River Irrigation Project Phase I
HVP = Hadejia Valley Project
Source: Barbier and Thompson (1998).

narios 1 and 1a. Given the high productivity of the floodplain, the losses in economic benefits due to changes in flood extent for all scenarios are large, ranging from US$2.6–4.2 million to US$23.4–24.0 million. As expected, there is a direct tradeoff between increasing irrigation upstream and impacts on the wetlands downstream. Scenario 3, which yields the lowest upstream irrigation gains, also has the least impact in terms of floodplain losses, whereas Scenario 5 has both the highest irrigation gains and floodplain losses. The results confirm that in all the scenarios simulated the additional value of production from large-scale irrigation schemes does not replace the lost production attributed to the wetlands downstream. Gains in irrigation values account for at most around 17% of the losses in floodplain benefits.

This combined hydrological-economic analysis would suggest that no new upstream developments should take place in addition to Tiga Dam. Moreover, a comparison of Scenario 3 to Scenario 2 in the analysis shows that it is economically worthwhile to reduce floodplain losses through releasing a substantial volume of water during the wet season, even though this would not allow Tiga Dam to supply the originally planned 27,000 ha on KRIP-I.

Although Scenario 3 is the preferred scenario, it is clearly unrealistic. As indicated above, Challawa Gorge was completed in 1992, and in recent years several

Table 2a. Impact of scenarios on mean peak flood extent and gains in total irrigated area.

	Scenario 1	Scenario 1a	Irrigated Area (km^2)
Scenario 2	–150.62	–211.20	270
Scenario 3	–95.25	–55.83	140
Scenario 4	–265.02	–325.60	270
Scenario 5	–870.49	–931.07	1 235
Scenario 6	–574.67	–635.25	220

Table 2b. Impact of scenarios in terms of losses in floodplain benefits versus gains in irrigated production, net present value (US$ 1989/90 prices).

		Scenario 1			Scenario 1a		
	Irrigation Value [1] a/	Floodplain Loss [2] b/	Net Loss [2] - [1]	[1] as % of [2]	Floodplain Loss [3] b/	Net Loss [3] - [1]	[1] as % of [3]
Scenario 2	682 983	-4 045 024	-3 362 041	16.88	-5 671 973	-4 988 990	12.04
Scenario 3	354 139	-2 558 051	-2 203 912	13.84	-4 184 999	-3 830 860	8.46
Scenario 4	682 963	-7 117 291	-6 434 328	9.60	-8 744 240	-8 061 277	7.81
Scenario 5	3 124 015	-23 377 302	-20 253 287	13.36	-24 004 251	-20 880 236	13.01
Scenario 6	556 505	-15 432 952	-14 876 447	3.61	-17 059 901	-16 503 396	3.26

Notes: a/ Based on the mean of the net present values of per hectare production benefits for the Kano River Irrigation Project Phase I (see Barbier & Thompson, 1998. Table 1), and applied to the gains in total irrigation area shown in Table 2a.
b/ Based on the mean of the net present values of total benefits for the Hadejia-Jama=are floodplain (see Barbier & Thompson, 1998. Table 1), averaged over the actual peak flood extent for the wetlands of 112,817 ha in 1989/90 and applied to the differences in mean peak flood extent shown in Table 2a.

Source: Barbier & Thompson (1998).

small dams have been built on the Hadejia=s tributaries while others are planned. Thus Scenario 4 most closely represents the current situation, and Scenario 5 is on the way to being implemented—although when the construction of Kafin Zaki Dam might occur is presently uncertain. As indicated in Table 2.b., full implementation of all the upstream dams and large-scale irrigation schemes would produce the greatest overall net losses, around US$20.2–20.9 million.

These results suggest that the expansion of the existing irrigation schemes within the river basin is effectively "uneconomic." The construction of Kafin Zaki Dam and extensive large-scale formal irrigation schemes within the Jama=are Valley do not represent the most appropriate developments for this part of the basin. If Kafin Zaki Dam were to be constructed and formal irrigation within the basin limited to its current extent, the introduction of a regulated flooding regime (Scenario 6) would reduce the scale of this negative balance substantially, to around US$15.4–16.5 million. The overall combined value of production from irrigation and the floodplain would, however, still fall well below the levels experienced if the additional upstream schemes were not constructed.

Such a regulated flooding regime could also produce additional economic benefits that are not captured in our analysis. Greater certainty over the timing and magnitude of the floods may enable farmers to adjust to the resulting reduction in the risks normally associated with floodplain farming. Enhanced dry season flows provided by the releases from Challawa Gorge and Kafin Zaki dams in Scenario 6 would also benefit farmers along the Hadejia and Jama'are Rivers while the floodplain's fisheries may also experience beneficial impacts from the greater extent of inundation remaining throughout the dry season. The introduction of a regulated flooding regime for the existing schemes within the basin may be the only realistic hope of minimizing floodplain losses. Proposed large-scale schemes, such as Kafin Zaki, should ideally be avoided if further floodplain losses are to be prevented. If this is not possible the designs for water resource schemes should enable the release of regulated floods in order to, at least partly, mitigate the loss of floodplain benefits that would inevitably result.

Currently, as a result of such economic and hydrological analyses of the downstream impacts of upstream water developments in the Hadejia-Jama'are floodplain both the states in northern Nigeria and the federal government have become interested in developing regulated flooding regimes for the upstream dams, and have been reconsidering the construction of Kafin Zaki Dam.

4. Upland Soil Erosion

At first glance, Case II as represented by equation (4) appears to be another straightforward, "unidirectional" externality problem. Farming in upper watersheds produces soil erosion, which leads to increased sedimentation in the watershed and thus affects downstream water supplies. This external impact is often referred to as the "off-site" costs of soil erosion in the literature. However, any erosion resulting from upland farming can also generate *on-site costs*, i.e., erosion of the topsoil may affect the long-run productivity of the soil and thus future returns from farming. Although the presence of such on-site costs suggests that upland farmers may have some incentive in reducing erosion on their land, there is no guarantee that this private rate of soil erosion will reflect the socially optimal rate of soil erosion. Upland farmers still have no incentive to consider the off-site costs of soil erosion, in terms of reduced water supplies downstream, in their erosion control decisions. By utilizing equation (4) in a simple watershed model of soil erosion and sedimentation, it is possible to show the conditions that determine the socially optimal, as opposed to the private, rate of upland erosion. A case study of upper watershed degradation in Java, Indonesia, further illustrates the valuation and incentive aspects of this externality problem.

Assume that farmers in the uplands are concerned with the effects of soil erosion on their discounted returns per hectare from agricultural production, V^3, which can be represented as follows:

$$V^3 = V^3(z_3, d, S, t;\ p^h, c, \delta) = \int_0^T (p^h h_3 - cz_3)\, e^{-\delta t}\ dt$$

$$\overline{S} = S_0 - d \tag{9}$$

$$h_3 = h_3(z_3, d, S), h_{3i} > 0, h_{3ii} < 0, i = z_3, d, S,$$

where upland agricultural yield, h_3, is an increasing function of topsoil depth, S, current erosion, d, and variable inputs, z_3. However, current erosion also reduces topsoil depth over time, and thus has a long-run impact on the productivity and returns of the upland system. This can be seen through examining the current value Hamiltonian, H', of the upland farmer's production decision, and some of the relevant first order conditions

$$H' = (p^h h_3 - cz_3) + \mu[S_0 - d]$$

$$\frac{\partial H'}{\partial d} = p^h h_{3d}(d^i) - \mu = 0$$

$$-\frac{\partial H'}{\partial S} = \dot{\mu} - \delta\mu = -p^h h_{3S} \text{ or } \frac{\dot{\mu}}{\mu} + \frac{h_{3S}}{h_{3d}} = \delta \quad \forall t, \tag{10}$$

where $\mu(t)$ is the imputed value, or "shadow price," of topsoil in terms of net returns to the upland farming system. By depleting topsoil through increased erosion, d, the upland farmer is essentially foregoing future returns to the system, as represented by $\mu(t)$. Consequently, the latter is the marginal *user cost* of additional soil erosion to the upland farming system. As condition (10) shows, the farmer determines the optimal private rate of soil erosion, d^i, by equating the marginal current returns to erosion with its imputed cost, $\mu(t)$. The latter is likely to be rising over time, as S is depleted.

Although the above model suggests that, under ideal conditions, the upland farmer is likely to internalize any user or on-site costs of soil erosion in production decisions, off-site costs will still be ignored. If the latter are significant, then the privately optimal rate of erosion determined by the upland farmer, d^i, will exceed the socially optimal rate, d^*. Once again, assuming a single river basin or watershed planning authority, this agency would be concerned with determining a rate of erosion that is compatible not only with maximizing the returns to upland farming, e.g., equation (9), but also with minimizing the negative off-site impacts on the downstream agroecosystem as well, e.g., equations (1) and (4). Thus the watershed authority would be interested in the following current value Hamiltonian of the social planning problem, and relevant first order conditions

$$H = (p^h h_1 - cz_1) + (p^h h_3 - cz_3) + \mu[S_0 - d] - \lambda g(d)$$

$$\frac{\partial H}{\partial d} = p^h h_{3d}(d^*) - \mu - \lambda g_d(d^*) = 0 \tag{11}$$

$$-\frac{\partial H}{\partial S}=\dot{\mu}-\delta\mu=-p^h h_{3S} \text{ or } \frac{\dot{\mu}}{\mu}+\frac{p^h h_{3S}}{[p^h h_{3d}-\lambda g_d]}=\delta$$

$$-\frac{\partial H}{\partial W}=\dot{\lambda}-\delta\lambda=-p^h h_{1W} \text{ or } \frac{\dot{\lambda}}{\lambda}+\frac{g_d p^h h_{1W}}{[p^h h_{3d}-\mu]}=\delta \quad \forall t,$$

where $\lambda(t)$ is the shadow price, or imputed value, of the downstream water supply as before. In comparison with (10), the watershed planning authority determines the optimal rate of erosion, d^*, by considering not just the on-site costs of erosion, μ, but also incorporates the off-site costs of sedimentation on downstream water supply and productivity, λg_d. As these costs are positive, then the socially optimal rate of erosion, d^*, will be less than the private rate, d^i, determined solely by upland farmers. Note, however, that if the downstream costs of sedimentation are fairly insignificant, then the private and social rates of erosion will coincide.

Because upland farmers have control over the process of erosion from their land, but may not necessarily have an economic interest in reducing erosion to socially optimal levels, this particular externality problem has its own unique set of valuation and incentive problems. These are illustrated through the following case study of upper watershed degradation in Java, Indonesia.

Case study: upper watershed degradation in Java, Indonesia

The following case study is based on Barbier (1989) and (1990), unless otherwise indicated.

On the densely populated island of Java in Indonesia, the area of severely eroded upland is increasing at the rate of 1–2% per annum and now covers a total of over 2 million hectares (ha), approximately one-third of Java's cultivated uplands. The population of the uplands is roughly 12 million. Population densities in these areas average 600–700 people per km^2, and farmland holdings average 0.4 ha or less. In some areas, up to 20–25% of the population are landless. Yields of upland rice and maize average 0.9 to 2.5 tons per ha. The general pattern is one of poor, predominantly subsistence households using inappropriate cropping patterns that result in high levels of soil erosion on their rain-fed lands. Significant cropland erosion is also caused by absentee and better-off farm owners who cultivate highly profitable but erosive crops such as vegetables on steep slopes with high water runoff.

Given these high rates of cropland erosion from upland farms in Java, the resulting on-site and off-site costs are likely to be substantial. In the late 1980s, a study attempted to estimate these costs (Magrath & Arens, 1989). On-site costs of upland erosion in Java in terms of lost crop productivity were estimated to be around US$327 million per annum, or 4% of the total dryland crop value in Java. The study estimated the off-site costs of sedimentation damage to lower watershed irrigation systems, reservoirs and harbors, which amounted to around US$25 to 91 million. Although other off-site effects may also occur, these could not be calculated.

These estimates of erosion costs are rough approximations. Nevertheless, they indicate that the off-site costs of soil erosion in Java may be significant, but relatively less important than the on-site costs. This in turn suggests that many upland farmers may have an incentive to reduce soil erosion on their cropland, as erosion is contributing to lost future productivity and income. Any resulting reduction in soil run-off and sedimentation could limit substantially the downstream costs of erosion as well. Thus encouraging farmers in the upper watersheds to reduce erosion will also benefit economic activities in the lower watershed that suffer from the downstream impacts of erosion.

On the other hand, the presence of significant on-site costs of soil erosion on cropland in the upper watersheds of Java also indicates that many upland farmers are not responding to these costs by automatically reducing erosion from their land. There appear to be several factors influencing upland farmers' decisions to ignore these costs.

For example, imperfect land and capital markets play a critical role in affecting the upland farmer=s decision to control soil erosion. The most reliable indicator that a farming household will have of the effects of soil erosion on future land productivity is through land prices. However, in many upland areas of Java, rural land markets are imperfect or distorted. Similarly, the lack of effective rural credit markets in the uplands may mean that the farmer has also to borrow in the short term to invest in land improvements. Thus distorted or non-existent local capital markets may make the costs of conservation prohibitively expensive. Throughout Indonesia, public liquidity credit meets only 15% of the demand for credit by farmers—and more than half of this is subsidized credit targeted at lowland sugar producers. Small farmers, particularly those outside of lowland irrigated areas, are dependent on informal credit markets, which can generate interest rates as high as 60%.

In addition, the relationship between the erodability and profitability of different farming systems on different soils and slopes across the upper watersheds of Java is an important determinant of whether upland farmers may adopt soil conservation measures. As illustrated in Figure 1, this relationship varies widely across Java. Two polar cases stand out. First, poorer upland farmers dependent on low-return cropping systems, such as maize and cassava, may be aware that soil erosion is reducing productivity but may not be able afford conservation measures. In particular, the adoption of terracing to control erosion on their sloping fields may be costly, but the result may hardly raise the overall profitability of the system. At the other extreme, rich farmers with very profitable crops that are extremely erosive, such as temperate vegetable crops and tobacco grown on steep upper volcanic slopes with deep topsoils, may face high rates of erosion. However, the short-term profits gained by these farmers are hardly affected by erosion, and any losses they face are more than compensated by generous production and input subsidies.

Both of these polar cases illustrate two of the most pervasive influences on upland farmers' attitudes to controlling soil erosion: the costs of traditional conservation "packages" available to upland farmers, and the role of agricultural policies in affecting the returns to different cropping systems.

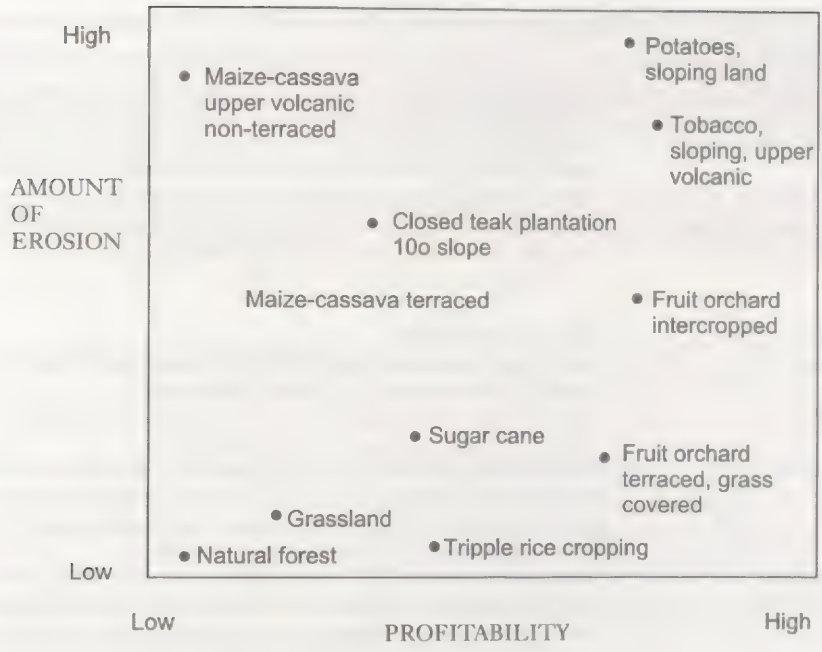


Figure 1. Adapted from Carson & Utomo (1986).

In the uplands of Java, the conventional soil conservation technology recommended for annual cropping on slopes up to 45–50% has been bench terracing, or back-sloping tied ridging. In the 1980s, a survey in the Citanduy watershed in West Java indicated that the total annual labor and material costs to farmers of constructing bench terraces could range from US$560 to US$2075 per ha (1979 prices). In comparison, Magrath and Arens (1989) estimated the average on-site cost of soil erosion in West Java to be only US$99 per ha. It is therefore not surprising that many farmers in the Citanduy watershed have failed to adopt bench terracing conservation, even when generous subsidies for construction were made available to them through soil conservation projects in this and other watersheds throughout Java.

To reverse these widespread disincentives, there is a need for a comprehensive rural development strategy for the Javan uplands that enhances farmers' incomes and productivity while at the same time reduces the economic pressures to degrade farmland. As Table 3 indicates, such a comprehensive policy and investment strategy to encourage sustainable agricultural development in the uplands of Java could easily be funded through reducing existing agricultural policy distortions and subsidies that are economically inefficient, fiscally damaging, and generally biased in favor of subsidizing richer farmers to engage in erosive practices. Reducing or eliminating input subsidies and reallocating research and extension funds could, in the short term, release US$245 million (1985 prices) annually for investment in sustainable agriculture in the uplands. Assuming a gradual phase-out of the fertilizer

Table 3. A comprehensive strategy for sustainable upland agricultural development in Java. (US$, 1985 prices)

Investment Strategy:	Source of Funds:
Short-Term Investment Strategy (US$245 million per annum)	US$150 million from a reduction in the fertilizer subsidy, US$25 million and US$40 million from the abolishment of the pesticide and sugar credit subsidies respectively, and US$30 million from the reallocation of research and extension funding.
Long-Term Investment Strategy (US$525 million per annum)	An additional US$190 million from the phase out of the fertilizer subsidy, and an additional US$90 million from a four-fold increase in the research and extension budget.
Annual Funding:	**Investment Program:**
US$35-40 million	Integrated pest management (IPM) for brown planthopper control, to be extended to IPM for other pests.
US$40-45 million	Increasing the availability of general rural credit to all upland farmers, at affordable rates and multiple terms.
US$60-240 million	Research and extension to develop and support new farming systems and land management techniques appropriate to upland conditions, including the development and dissemination of HYVs appropriate to diversified upland systems.
US$110-200 million	Further investments in improved farming systems for specific upland agroecological zones, and in improvements in the physical infrastructure serving these zones, including rural transport, integration of markets, credit facilities, post-harvest technology and processing, and produce quality.

Source: Adapted from Barbier (1989).

subsidy and a fourfold increase in both research and extension budgets, this could increase to as much as US$525 million per year (1985 prices). This policy should be complemented by the removal of effective protection for vegetables, sugar, and other crops that are encouraging richer upland farmers to degrade land for short-term profits.

Although the government of Indonesia has reduced the pesticide subsidy in recent years, there has been little change in its general agricultural and credit policy. Sadly, Indonesia's current economic crisis has meant that the opportunity to redirect this strategy toward more sustainable upland development has been lost for the forseeable future. Thus upland land degradation will continue, and with it, substantial downstream economic costs.

5. Upper Catchment Timber Harvesting

This final case differs from the first two as it concerns a "positive" externality between the upper and lower watershed. That is, as indicated by equation (5), timber harvesting in the upper watershed may actually increase the overall water flow through the catchment, thus augmenting the amount of water available downstream. However, timber operators in the upper watershed have little incentive to consider these additional benefits to downstream users of water. By including (5) in a catchment-wide model, it is possible to determine the optimal rate of

water-yield augmentation through timber harvesting in the uplands, and compare it to the private harvesting decision of operators. The valuation and incentive issues arising from this type of ecological and economic externality are further explored in a case study of upper catchment forest management for both increased timber and water yields in the Colorado River Basin.

A timber operator in the upper catchment is concerned with the discounted returns, V^4, from harvesting timber, which can be represented as follows

$$V^4 = V^4(z_4, F, t;\ p^v, c, \delta) = \int_0^T (p^q q - cz_4)\, e^{-\delta t}\ dt$$

$$\dot{F} = G(F) - q \qquad (12)$$

$$q \quad \alpha z_4 F,$$

where timber yield, q, is proportional to the application of variable inputs, z_4, to the stock of timber resources per hectare, F. Net growth in the production forestry stock, dF/dt, is therefore determined by biological growth, $G(F)$, net of harvest, q. Thus the timber operator is interested in managing forests so as to maximize the returns from harvesting timber only, which suggests the following current value Hamiltonian and relevant first-order conditions

$$H' = p^q q - \frac{cq}{\alpha F} + \sigma[G(F) - q]$$

$$\frac{\partial H'}{\partial q} = p^q - \frac{c}{\alpha F} - \sigma = 0$$

$$-\frac{\partial H'}{\partial F} = \dot{\sigma} - \delta\sigma = -\frac{cq^i}{\alpha F^2} - \sigma G_F \text{ or } \frac{\dot{\sigma}}{\sigma} + \frac{cq^i}{\sigma\alpha F^2} + G_F = \delta, \quad \forall t,$$

where $\sigma(t)$ is the imputed value, or "shadow price," of upland production forests in terms of net returns to the upper catchment forestry system. By harvesting standing timber today rather than maintaining it, the timber operator is essentially foregoing future returns to the system, as represented by $\sigma(t)$. As condition (13) indicates, the optimal private rate of timber harvesting, q^i, occurs when marginal current returns to harvesting just equal this imputed cost, $\sigma(t)$.

However, it is clear that the upper catchment timber operator has no incentive to manage upland forests to yield both timber and water benefits. Thus the privately optimal level of harvest, q^i, preferred by the operator is unlikely to be the same as the socially optimal harvest, q^*, that considers the water-yield impacts of upper catchment timber extraction as well. Thus, again assuming a single catchment planning authority, this agency would be concerned with determining a rate of harvest that is compatible not only with maximizing the returns to upper catchment forestry operations, e.g., equation (10), but also with maximizing the positive off-site impacts on the downstream agroecosystem as well, e.g., equations (1) and (5). Thus the watershed authority would be interested in the following current value Hamiltonian of the social planning problem, and relevant first-order conditions

$$
\begin{aligned}
&H = (p^h h_1 - cz_1) + (p^q q - \frac{cq}{\alpha F}) + \sigma[G(F) - q] + \lambda d(q) \\
&\frac{\partial H}{\partial q} = p^q - \frac{c}{\alpha F} - \sigma + \lambda d_q(q^*) = 0 \\
&-\frac{\partial H}{\partial F} = \overline{\sigma} - \delta\sigma = -\frac{cq^*}{\alpha F^2} - \sigma G_F \text{ or } \frac{\overline{\sigma}}{\sigma} + \frac{cq^*}{\sigma\alpha F^2} + G_F = \delta, \\
&-\frac{\partial H}{\partial W} = \overline{\lambda} - \sigma\lambda = - p^h h_{1W} \text{ or } \frac{\overline{\lambda}}{\lambda} + \frac{p^h h_{1W}}{\lambda} = \delta \quad \forall t,
\end{aligned}
\tag{13}
$$

where as before $\lambda(t)$ is the shadow price, or imputed value, of the downstream water supply in terms of additional returns to the lower catchment agroecosystem. In comparison with (13), condition (14) suggests that the upper catchment timber operator should harvest more timber each period, as there is an additional benefit of harvesting, λd_q. However, because these latter benefits accrue to the downstream agroecosystem and not to the upstream forestry system, the timber operator does not have an incentive to increase profit-maximizing harvest levels, h', to the socially optimal levels, h , that yield both timber and water benefits.

Consequently, the problem of valuing the water-yield augmenting externality is also connected to determining the appropriate incentives for encouraging upper catchment forestry management to consider both timber and water production benefits in harvesting decisions. The following case study of upper catchment forest management for both increased timber and water yields in the Colorado River Basin examines some of these valuation and incentive issues further.

Case study: joint production of timber and water yields in the Colorado River basin, U.S.A.

The following case study is based on Bowes and Krutilla (1989).

Existing subalpine forests in the central Rockies of the United States are considered to be of low quality for timber production. It has been suggested that replacing some of these forests with managed stands of better quality timber, could boost harvest yields substantially. A harvesting pattern of gradual removal of existing trees, followed by natural regeneration, periodic thinning, and final harvest could be instigated and repeated over multiple rotations. Moreover, if the pattern of tree removal in the snow zones of the central Rockies leads to small openings that improve the efficiency of the snowmelt, then the harvesting rotation could also lead to augmentation of the water yield from the subalpine snowpack. As the increased water flowing into the Colorado River Basin is not from storm flows but rather from the melting winter snowpack, there is unlikely to be any impact on downstream flood peaks. In addition, the anticipated increase in the volume of river flow (estimated to be 3% at its maximum) is anticipated to have little impact on the immense storage capacity of the Colorado River. Thus there is not expected to be any flooding problems associated with the increased water yield from timber harvesting.

It is therefore possible to design a forest harvesting strategy that yields joint timber and water yield benefits. Any resulting water augmentation to the Colorado River Basin will most likely lead to additional withdrawals for industrial and municipal use in the Front Range states, in particular the Denver-Fort Collins region of Colorado. Increased water yields may also occur on the West Slope and in Lower Colorado Basin states, notably Arizona, Nevada, and California, which would benefit hydroelectric, irrigation, municipal, and industrial uses.

Table 4 depicts the results of the analysis by Bowes and Krutilla (1989) for the joint production of timber and water benefits for the Fraser-Colorado River Basin. There are several possible scenarios that could generate potential timber and water yield benefits. The most conservative scenario (Scenario A) is just economically worthwhile with a 4% discount rate. At the other extreme, Scenario E offers the highest water augmentation benefits and overall net present value, but timber is produced at an economic loss. However, the results of Table 4 are very sensitive to roading costs, and hence the need for greater road distances, construction on steeper slopes or more permanent roads may make exploitation of sub-alpine

Table 4. Present value water and timber benefits, Colorado-Fraser River Basin. (US$ per acre, 4% discount rate) 1/

Scenario Rotation age/	Rotations age/ Entry interval	Timber value	Water value 2/	Roading costs	Total NPV
A	120/30	$ 23	$238	$237	$ 24
B	120/30	$ 30	$295	$237	$ 88
C	90/30	$ 36	$298	$237	$ 97
D	60/20	$ 53	$332	$263	$122
E	30/30	$ -1	$390	$237	$152

Scenario A: At initial entry, one third of original forest would be patch-cut harvested; the next third patch-cut harvested in 30 years, and first stand thinned; and the final third patch-cut in 60 years, with thinning on other two sites. Thinnings occur on all regenerated stands at 30-year intervals until final harvests at age 120 for each stand.

Scenario B: Same as A, except at initial entry residual stands are thinned to half their density.

Scenario C: Same as B, except final harvests for each regenerated stand occurs at age 90.

Scenario D: At initial entry, one third of original forest would be patch-cut harvested; the next third patch-cut harvested in 20 years, and first stand thinned; and the final third patch-cut in 40 years, with thinning on other two sites. One additional thinning precedes the final harvest at age 60 for each regenerated stand. The more frequent re-entry interval increases roading costs.

Scenario E: At initial entry, half the original forest is harvested; remaining half is thinned to half its density but not re-entered in later years. Reharvesting of regenerated half of forest occurs at 30-year rotations.

Notes: 1/ Assumes moderate slope, seasonal roads (permanent construction, open except for seasonally bad conditions), and 5.5 miles of access road per square mile. The US Office of Management and Budget authorizes a 4% real rate of interest for timber operations of the US Forest Service.

2/ Based on the following marginal values: hydroelectricity ($0.018/kilowatt-hour), Upper Basin irrigation ($10/acre-feet), Upper Basin municipal and industrial use ($60/acre-feet), Lower Basin municipal and industrial use ($100/acre-feet).

Source: Adapted from Bowes & Krutilla (1989).

forests uneconomic. In addition, an increase in the discount rate to 7% also excludes more potential sites from the harvesting strategy.

The choice of timber operation strategy also raises the issue of incentives. The sub-alpine forests of the central Rockies are essentially public forestlands managed by the US Forest Service. Traditionally, the main interest of the Forest Service is in the production of timber values, and not increased water yields *per se*. As noted by Bowes and Krutilla (1989), it is not evident that the current method of budgeting and funding Forest Service programs through the Office of Management and Budget (OMB) and approval by Congress is sufficiently flexible to allow the Forest Service to develop appropriate multi-use forest management strategies, such as the joint production of timber and water. As Table 4 indicates, timber values alone are insufficient to justify any of the harvesting scenarios.

One possibility, advocated by Bowes and Krutilla (1989), is reform of the financing process so that the Regional Offices of the US Forest Service and OMB work more closely to identify funding priorities for key multi-purpose management projects, such as the joint production of timber and water yields in the Colorado River Basin. However, a more innovative approach would be for major downstream users, such as the Denver and Fort Collins municipalities, electrical utilities, or even individual state governments, to collaborate with the US Forest Service on funding the water yield component of the joint production harvesting strategy in the central Rockies. Such an approach to watershed management to secure improved downstream services is not without precedence in the United States. As noted by Chichilnisky and Heal (1998), in 1996 New York City began investing between US$1 to 1.5 billion in protecting the natural water purification process of the Catskill watershed—by buying land and subsidizing the construction of better sewage treatment in the upper watershed. However, the expected gain to New York City is that they will not have to invest US$6 to 8 billion, plus running costs of US$300 million annually, on a filtration plant to provide cleaner water to the metropolitan area.

6. Conclusion

This paper has considered three important ecological and economic externalities associated with the provision of water through watersheds and drainage basins. Each type of externality has raised specific valuation and incentive issues. In the case of upstream water diversion, the decision to divert additional water upstream must take into account the impacts on economic activities dependent on the use of water in the lower watershed and river basin. With soil erosion from upland cropping systems, a critical issue is the significance of productivity and income losses to these farms. Where these are substantially greater than off-site costs, then improving the incentives for farmers to invest in better farming systems and land management may also reduce a large component of the costs of sedimentation to downstream economic activities. Finally, the joint management of forestlands for timber and water yields will benefit water users in the lower watershed, but a criti-

cal issue is ensuring that this contribution is accounted for in the choice of forest management strategy in the upper watershed.

As the examples in this paper indicate, valuation issues cannot be separated from the need to design effective economic strategies and appropriate incentives. Also, analysis of the ecological externality problem is directly relevant to understanding the critical economic externality involved. Although this paper has used examples from watershed and river basin management to illustrate these principles, they are generally applicable to the many other cases where external ecological support functions also have important economic values.

Acknowledgments

The author wishes to acknowledge the helpful comments of two anonymous referees. However, the usual caveat applies.

Literature Cited

Aylward, B. A. & E. B. Barbier. 1992. Valuing environmental functions in developing Countries. Biodiversity and Conservation 1: 34–50.

Barbier, E. B. 1989. Economics, Natural Resource Scarcity and Development: Conventional and Alternative Views. Earthscan, London.

Barbier, E. B. 1990. The farm-level economics of soil erosion: The uplands of Java. Land Economics 66(2): 199–211.

Barbier, E. B. 1994. Valuing environmental functions: Tropical wetlands. Land Economics 70(2): 155–173.

Barbier, E. B. & J. R. Thompson. 1998. The value of water: Floodplain versus large-scale irrigation benefits in northern Nigeria. Ambio 27(6): 434–440.

Barbier, E. B., W. M. Adams & K. Kimmage. 1993. An economic valuation of wetland benefits. Pp. 191–209 *in* G. E. Hollis, W. M. Adams & M. Aminu-Kano (editors), The Hadejia-Nguru Wetlands: Environment, Economy and Sustainable Development of a Sahelian Floodplain Wetland. IUCN, Gland, Switzerland.

Barbier, E. B., J. C. Burgess, J. T. Bishop & B. A. Aylward. 1994. The Economics of the Tropical Timber Trade. Earthscan, London.

Beeby, A. 1993. Applied Ecology. Chapman and Hall, London.

Bowes, M. D. & J. V. Krutilla. 1989. Multiple-Use Management: The Economics of Public Forest Lands. Resources for the Future, Washington, D.C.

Calish, S., R. D. Fight & D. E. Teeguarden. 1978. How do nontimber values affect Douglas-fir rotations? J. Forestry 76: 217–221.

Carson, B. & W. H. Utomo. 1986. Erosion and sedimentation processes in Java. KEPAS, Malang, Indonesia.

Chichilnisky, G. & G. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–630.

Cornes, R. & T. Sandler. 1986. The Theory of Externalities, Public Goods and Club Goods. Cambridge Univ. Press, Cambridge.

Freeman, A. M., III. 1993. The Measurement of Environmental and Resource Values: Theory and Methods. Resources for the Future, Washington, D.C.

Magrath, W. & P. Arens. 1989. The Costs of Soil Erosion on Java: A Natural Resource Accounting Approach. Environment Department Working Paper No. 18, The World Bank, Washington, D.C.

Wilcock, D. N. 1993. River and inland water environments. Pp. 59–91 *in* B. Nath, L. Hens, P. Compton & D. Devuyst (editors), Environmental Management: Volume II The Ecosystems Approach. VUB Univ. Press, Brussels.

The Monetary Valuation of Biodiversity: Promises, Pitfalls, and Rays of Hope

John M. Gowdy
Professor of Economics and Director
of the Ph.D. program in Ecological Economics
Rensselaer Polytechnic Institute
Troy, New York 12180, U.S.A.

Abstract

Markets have proven to be efficient allocators of resource inputs into the economic process. For most economic goods, price signals function quite well in eliminating surplus and shortages. It is only natural, then, that economists apply the principles of market economies to environmental problems like biodiversity loss and global climate change. And indeed, there are many examples of the successful use of the price mechanism to alleviate particular environmental problems. There is, however, a real danger in placing monetary values on the essential workings of nature upon which all life depends. Although market exchange values of environmental services may be used to justify measures to protect biodiversity, it must be stressed that exchange value constitutes a small portion of the total value of any particular feature of the biological world. Unresolved, and perhaps unresolvable, issues in environmental valuation include discounting, the concept of marginal value, and the economic notion of sustainability.

Introduction

The past few months have seen a number of well-publicized efforts to place monetary values on nature's services. A study by Costanza et al. (1997) valuing the services of the World's ecosystems received extensive media coverage and has already spurred similar studies at the level of regions and ecosystems. A recent

issue of *Science* was devoted to the theme of "Human-Dominated Ecosystems." Books by Yvonne Baskin (1997) and Gretchen Daily (1997) have received widespread attention and have done much to increase public awareness of the intricacy of ecosystems and their value to humanity.

There are many positive examples of market forces leading to biological conservation. The government of Bermuda, for example, has paid fishermen not to fish on its beautiful but deteriorating coral reefs, because the reefs are a source of millions of dollars of tourist income each year (Pennisi, 1997). The New York City watershed agreement, wherein the City of New York agreed to pay a large sum of money to residents of the Catskills in order to protect the City's water supply, is another indication that society is beginning to recognize the economic value of ecosystems (Chichilnisky & Heal, 1998; Lubchenco, 1998). The Merck Pharmaceutical Company agreed to pay The National Institute of Biodiversity in Costa Rica $1 million for the right to prospect for rainforest drugs (Myers, 1997). Efforts like these are encouraging and absolutely necessary if the crisis of biodiversity loss is to be addressed. Geoffrey Heal (1997) is certainly right when he says that the higher the market value placed on nature's services the more likely they will be preserved.

The modern market system has brought humankind unbelievable technological achievements and has made life comfortable and secure for a substantial portion of the Earth's population. Markets have proven to be so successful as a way of organizing economic production, it appears that no viable alternatives now exist. In the socio-economic system we have created, price signals have performed quite well in allocating resources, forcing technological advances when they are most needed, and in general helping us to avoid economic scarcities of natural resources. Within this context, it seems natural to us to respond to biological scarcities as we would to economic scarcity. That is, if "too much" biodiversity is being used up then increase its price to reflect its "true" value.

As many economists and biologists have pointed out, however, there is a real danger in placing monetary values on the essential workings of nature on which all life depends. Although market exchange values of environmental services may be used to justify measures to protect biodiversity, it must be stressed that exchange value constitutes a small portion of the total value of any feature of the biological world. Consider a simple thought experiment. Costanza et al. (1997) estimated that the total economic value of the Earth's environmental services is $33 trillion. For any market exchange to take place there must be both a willing buyer and a willing seller. Would any sane person accept the offer of $33 trillion for planet Earth? Or for that matter, an offer for any amount of money at all. Certainly not, because life itself would end, and this surely says something about the ultimate folly of valuing nature by the same yardstick that we value easily reproducible market goods. Following Anderson (1966) and Brown (1984), economic measures of value should be considered to be species of the genus *assigned value*, which belongs to the family *value*. When considering the value of nature's services, biodiversity, for example, all the various hierarchies of value, market value, value to human soci-

ety, and value to ecosystem functioning must be, in one form or another, taken into account.

Tilman (1997: 93) defined biodiversity as "the variety of life at all levels of organization, from the level of genetic variation within and among species to the level of variation within and among ecosystems and biomes." As Tilman and others have demonstrated, increasing biodiversity, up to a point, is correlated with increasing biomass and increasing resilience to disturbance. For a particular ecosystem it is possible to construct a quantitative measure of biodiversity. In some cases, however, increases in biological productivity may be correlated with a reduction in biodiversity (Rosenzweig, 1971; Tilman, 1982). In contrast to market-based measures of the value of biodiversity, which reduce all biological features to a single, universal metric (money), biological measure are place- and time-specific (Magurran, 1988).

The foundation of modern economics is the theory of the consumer, what economists call utility theory. This theory has been developed over the past 200 years with contributions from moral philosophy, physics, mathematics, and even law. The theoretical edifice of neoclassical economics is undeniably one of the crowning intellectual achievements of our age. The focus of neoclassical theory, however, is very limited, namely, the rules that govern exchange in a competitive market economy. The methods and assumptions of utility theory underlie attempts to place market values on the world's natural features. It is important that the economists who make these estimates and the biologists who use them understand clearly the limitations of the underlying analysis, and the limitations of market allocation of nature's services.

Neoclassical economics is a description of market exchange pared down to its simplest features. Like any model, the theory attempts to describe the essential features of its subject matter with the fewest assumptions. And given this, one must say that the model captures the essential features of a market economy. Many criticisms of neoclassical theory are better directed to the market system economic theory describes. Features that work against the preservation of the natural world, such as discounting the future, marginal valuation, and "value monism," are features of real market economies, not the figments of economists' imaginations. Economic theory shows quite clearly why markets are incapable of capturing all the value of nature or of insuring environmental sustainability. As Bishop (1992) pointed out, the problem is that economists have been lazy in following through on the implications of the theory.

Issues in Market Valuation of the Natural World

Discounting the future

To be used in economic analysis, estimates of the future value of goods and services, including the services of nature, must be discounted. Market decisions are

driven by individual decisions made at a particular point in time. It is natural that an individual, faced with pure uncertainty and a finite life span, should prefer something today rather than at some time in the future. The neoclassical view is that individual preferences should be taken as the starting point for economic analysis and that these preferences can be correctly revealed in market outcomes. In this view, individuals are rational and are the best judge of what is good for them, the choices that give individuals the greatest amount of utility are those revealed to be preferred in market choices, and these market outcomes are sacrosanct. According to the standard approach, the principle of consumer sovereignty compels us to accept people's tastes including their intertemporal preferences.

A major problem with this position is that fundamental differences exist between the choices individuals make in the context of markets, and the choices they make in a larger context as members of a community of other humans. Price (1993: 102–103) wrote:

> There is a fatal weakness in the parallel between contemporary and intertemporal choice. Consumers may choose between two goods at one point in time on an equal footing (if similar information is given about each): they do not so choose between two time periods, for the choice is made *from within one of those time periods*. This asymmetry may make irrationality and misinformation more serious in intertemporal choice than in choice between contemporary acts of consumption.

The shortcomings of discounting, whether it is used in private market decisions or in public policy using cost-benefit calculations, are nowhere more serious than in the case of biodiversity preservation. Economically rational decisions made in a market context by individuals at a specific point in time may collectively result in the loss of a resource critical to the long-term survival of the human species. For example, the present "value of everything," $33 trillion, delivered in 100 years, discounted at a relatively low rate of 5% would be about $251 billion, a sum less than the economic output of Michigan in 1996 (U.S. Bureau of the Census, 1998).

Marginal value

Central to economic theory is the concept of marginal value. It is a concept so ingrained in how economists think about markets and values, it is almost impossible to appreciate its limits. When we consider the value of a species or an ecosystem, the concept of marginal value is problematic. Removing or adding one species will affect all the others in the system in largely unpredictable ways. Biodiversity is characterized by "functional transparency" (Vatn & Bromley, 1994). The contribution of one feature of an ecosystem cannot be known until it is added or subtracted to the system. And furthermore, the effect is likely to be different each time a change is made. There are also some unresolved issues in neoclassical utility theory that call into question the universal applicability of the

notion of marginal value. The existence of lexicographic ordering of preferences, that is, some goods and services are considered to be absolutely essential and not subject to trade (Georgescu-Roegen, 1936, 1950), has been empirically verified in a number of studies of consumer preferences (see, for example, Spash & Hanley, 1995).

Neoclassical utility theory, with its underlying assumptions of non-satiation, universal substitutability, and diminishing marginal utility, is the basis of economic measures of value including the value of biodiversity. It is important for biologists to understand that there are many basic unresolved questions in utility theory. For example, economists assert that "utility" does not require a cardinal measure, that is, it is not necessary to be able to assign specific values (or "utils") to collections of commodities. All that is needed is the ability to consistently rank commodity bundles according to which are more and which are less preferred. Georgescu-Roegen (1968) pointed out, however, that diminishing marginal utility has no meaning without some notion of cardinality. It is not enough to be able to merely rank goods according to the utility they give. To construct a complete preference ordering we need to be able to rank not only commodity bundles but also the difference between them (weak cardinality). We must also reduce the value of every desired commodity to a single common denominator, its utility, or taking it a step further, its price. In market value, money becomes a substitute for anything. Can one imagine a single biological measure that could be used to compare the "values" of species or ecosystems anywhere on the planet at any time? So the question "What is the marginal value of a species?" is fraught with conceptual as well as informational difficulties.

Weak Sustainability

Basic to neoclassical economics is the sanctity of individual choice, the fungibility of economic goods and inputs, and faith in the market system to bring forth substitutes as relative prices change. These notions are part of what Joseph Schumpeter called the "pre-analytic vision" of neoclassical economics (Daly, 1995), and they are central to the concept of weak sustainability, a topic of intense debate among ecological economists (Gowdy, 1997; Gowdy & O'Hara, 1997; Pearce & Atkinson, 1993; Tisdell, 1997; Victor, 1991). Weak sustainability stipulates that an economy is sustainable if its capacity to generate income for future generations, a capacity embodied in its capital stock, is maintained (Hartwick, 1978; Solow, 1991). The assumption of substitutability among different kinds of capital is central to weak sustainability. If natural capital (non-renewable resources and ecological services, Folke et al., 1994), manufactured capital, and human capital are substitutes, it is permissible to focus only on total capital stock. Weak sustainability is achieved if an economy saves more than the combined depreciation of the various kinds of capital, even if it draws down its stock of natural resources (Pearce & Atkinson, 1993).

The neoclassical economic meaning of sustainability is sustaining the growth rate of Net National Product (Gross Domestic Product minus capital depreciation). GDP may be seen as the rate of return on capital stock, so sustainability is basically a problem of managing a nation's portfolio of capital stock.

The economic perspective on sustainability, then, is from the point of view of an individual-acting-at-a-point-in-time, nation-wide or even planet-wide portfolio manager. Weak sustainability extends the economic problem of the individual consumer of economic theory, *Homo economicus*, from maximizing utility subject to a budget constraint to maximizing economic output subject to the limited amount of total capital. The starting point then for the economic view of sustainability is the notion of economic man: (1) *Homo economicus* is an isolated individual entering an impersonal market at a particular point in time, (2) the goal of *H. economicus* is to maximize utility, (3) utility is derived from consuming goods and services, and (4) the value of goods and services is accurately reflected in their market prices. It follows that the larger Gross Domestic Product, the greater is total utility. From this logic it follows that the goal of sustainability is to insure that GDP is non-declining in the future.

An instructive example of weak sustainability in practice is the small Pacific island nation of Nauru (Gowdy & McDaniel, 1995, 1999; McDaniel & Gowdy, 2000). In 1900 one of the world's richest phosphate deposits was discovered on Nauru and today, as a result of just over 90 years of phosphate mining, about 80% of the island is totally devastated. It is impossible to even walk around in the mined-out areas. The island was self-sufficient before phosphate mining, but now almost all food and even drinking water must be imported. The substitution of imported canned foods for local fresh products, the increased consumption of alcohol, and the adoption of other patterns of an industrialized-world lifestyle, have adversely affected the health of the people of Nauru. Nearly 30% of those over 25 and about 50% of those over 50 have diabetes (Van Atta, 1997). Very high rates of heart disease and hypertension have also been reported. Life expectancy at birth for males in 1985 was 49 years, substantially less than that of other Pacific Islanders except for Papua New Guinea males, which was also 49 years (Taylor & Thoma, 1985). The island is biologically impoverished, chances of rehabilitation are remote, and there is no way the island could support its current population of 10,000, or even its pre-European contact population of about 1200, without a steady flow of goods from the outside world.

At the same time, the people of Nauru have had, over the past several decades, a high per capita income. Income from phosphate mining has enabled the Nauruans to establish a trust fund estimated to be as large as $1 billion. Interest from this trust fund should insure a substantial and steady income and thus the economic sustainability of the island. Nauru has traded its physical environment for money and increased its income by doing so. Nauru has followed the logic of weak sustainability, destroying its resource base and converting it into financial capital. The Nauru case shows clearly that weak sustainability may be consistent with a situation of near-complete environmental devastation.

Rays of Hope

A growing number of economists argue that the basic problem with neoclassical economics is its exclusive emphasis on a very restrictive theory of individual choice. A number of seemingly unrelated "movements" will eventually, I believe, change the way the economics profession views the economy and the natural world. All of these will challenge basic economic notions of how to value nature's attributes.

- Re-examination of the foundations of utility theory—There is an exponentially growing literature exploring the philosophical, psychological, and ethical foundations of neoclassical theory (Dasgupta, 1998; Bromley, 1998). New ideas broadening the economic concept of value are beginning to show up in mainstream economic journals.
- Re-examination of the "microfoundations" controversy—The past 20 years or so has seen the ascendancy of the view that the macro economy can be explained by the same principles that govern economic behavior at the micro level (as in the weak sustainability concept). This view has always been controversial within the profession, but the challenges now seem to be on a firmer footing and are being reinforced by the growing evidence of the negative effects of economic activity on the Earth's atmospheric, biological, and geochemical systems (Bromley, 1989; Colander, 1996; Hodgson, 1993).
- Re-capturing the commons—There has been an explosion of literature in recent years challenging the sanctity of individual choice and arguing for new institutional arrangements protecting future generations and environmental integrity (Bromley, 1989; Norton et al., 1998; Sen, 1993).
- Corporate responsibility—There are encouraging signs that the corporate world is moving away from Friedman's position that the only social responsibility of the firm is to maximize profits. Corporations such as Monsanto and British Petroleum have taken leading roles, not only in promoting sustainable production practices, but also in addressing the major environmental issues of our time: biodiversity loss and global climate change.

Final Thought

As Jane Lubchenco (1998) so eloquently argued in her outgoing address as president of the American Association for the Advancement of Science, there is an increased realization of the connections between the economy, the biosphere, and human social systems. A new social contract is needed committing scientists to work toward solving the pressing problems we face. Solving these problems will require unprecedented interdisciplinary cooperation and understanding. To be successful, interdisciplinary work requires more than merely assembling specialists from a variety of relevant disciplines. It requires that each discipline's practitioners have

some understanding of other relevant subjects. Economists must learn the rules by which ecosystems operate, and biologists must learn the rules by which markets operate. It should not be assumed, as is too often the case, that these rules are in harmony with each other. Biologists are perhaps in a better position than economists to understand this.

Literature Cited

Anderson, B. 1966. Social Value. Augustus M. Kelley, New York.

Baskin, Y. 1997. The Work of Nature. Island Press, Washington, D.C.

Bishop, R. 1992. Economic efficiency, sustainability, and biodiversity. Ambio 22: 69–73.

Bromley, D. 1989. Economic Interests and Institutions: The Conceptual Foundations of Public Policy. Blackwell, Oxford, U.K.

Bromley, D. 1998. Searching for sustainability: The poverty of spontaneous order. Ecological Economics 24: 231–240.

Brown, T. 1984. The concept of value in resource allocation. Land Economics 60: 231–246.

Chichilnisky, G. & G. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–630.

Colander, D. 1996. Beyond Microfoundations. Cambridge Univ. Press, New York.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. J. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Daily, G. (Editor). 1997. Nature's Services. Island Press, Washington, D.C.

Daly, H. 1995. Review of Wilfred Beckerman, Small is Stupid: Blowing the Whistle on the Greens. *In* Population and Development Review, 665–673.

Dasgupta, P. 1998. Population, consumption and resources: Ethical issues. Ecological Economics 24: 139–152.

Folke, C., M. Hammer, R. Costanza & A. Jansson. 1994. Investing in natural capital—Why, what, and how? *In* C. Folke, M. Hammer, R. Costanza & A. Jansson (editors), Investing in Natural Capital. Island Press, Washington, D.C.

Georgescu-Roegen, N. 1936. The pure theory of consumer's behavior. Quarterly Journal of Economics 50(3): 545–593.

Georgescu-Roegen, N. 1950. The Theory of Choice and the constancy of economic laws. Quarterly Journal of Economics 64(1): 125–138.

Georgescu-Roegen, N. 1968. Utility. *In* The International Encyclopedia of the Social Sciences. MacMillan, New York.

Gowdy, J. 1997. The value of biodiversity: Markets, society, and ecosystems. Land Economics 73(1): 25–41.

Gowdy, J. & C. McDaniel. 1995. One world, one experiment: Addressing the economics-biology conflict. Ecological Economics 14(3): 180–192.

Gowdy, J. & C. McDaniel. 1999. The physical destruction of Nauru: An example of weak sustainability. Land Economics 75(2): 333–338.

Gowdy, J. & S. O'Hara. 1997. Weak sustainability and viable technologies. Ecological Economics 22(3): 239–248.

Hartwick, J. 1978. Substitution among exhaustible resources and intergenerational equity. Review of Economic Studies 54: 347–543.

Heal, G. 1997. Valuing Our Future: Cost-Benefit Analysis and Sustainability. Program on Information and Resources, Columbia University, New York.

Hodgson, G. 1993. Economics and Evolution. Univ. Michigan Press, Ann Arbor.

Lubchenco, J. 1998. Entering the century of the environment: A new social contract for science. Science 279: 491–497.

Magurran, A. 1988. Ecological Diversity and Its Measurement. Princeton Univ. Press, Princeton, New Jersey.

McDaniel, C. & J. Gowdy. 2000. Paradise for Sale: A Parable of Nature. Univ. California Press, Berkeley.

Myers, N. 1997. Biodiversity's genetic library. Pp. 255–274 *in* G. Daily (editor), Nature's Services. Island Press, Washington, D.C.

Norton, B., R. Costanza & R. Bishop. 1998. The evolution of preferences: Why "sovereign" preferences may not lead to sustainable policies and what to do about it. Ecological Economics 24: 193–211.

Pearce, D. W. & G. D. Atkinson. 1993. Capital theory and the measurement of sustainable development: An indicator of "weak" sustainability. Ecological Economics 8: 103–108.

Pennisi, E. 1997. Brighter prospects for the world's coral reefs? Science 277: 491–493.

Price, C. 1993. Time, Discounting and Value. Blackwell, Oxford, U.K.

Rosenzweig, M. 1971. Paradox of enrichment: Destabilization of exploitation ecosystems in ecological time. Science 171: 385–387.

Sen, A. 1993. Markets and freedoms: Achievements and limitations of the market mechanism in promoting individual freedoms. Oxford Economic Papers 45: 519–541.

Spash, C. & N. Hanley. 1995. Preferences, information, and biodiversity preservation. Ecological Economics 12(3): 191–208.

Solow, R. 1991. Sustainability: An economist's perspective. *In* R. Dorfman & N. Dorfman (editors), Economics of the Environment, 3rd ed. W. W. Norton, New York.

Taylor, R. & K. Thoma. 1985. Mortality patterns in the modernized Pacific island nation of Nauru. American Journal of Public Health 75: 149–155.

Tilman, D. 1982. Resource Competition and Community Structure. Princeton Univ. Press, Princeton, New Jersey.

Tilman, D. 1997. Biodiversity and ecosystem functioning. *In* G. Daily (editor), Nature's Services. Island Press, Washington, D.C.

Tisdell, C. 1997. Weak and strong conditions for sustainable development: Concepts and policy implications. Wissenschaftszentrum Berlin, FS II 98-402.

U.S. Bureau of the Census, Statistical Abstract of the United States: 1998 (118th ed.), Table 719, p. 454, Washington, D.C.

Van Atta, D. 1997. Paradise squandered. Readers Digest May: 87–91.

Vatn, A. & D. Bromely. 1994. Choices without prices without apologies. Journal of Environmental Economics and Management 26: 129–148.

Victor, P. A. 1991. Indicators of sustainable development: Some lessons from Capital theory. Ecological Economics 4: 191–213.

Paths to Sustainable Development: The Knowledge Revolution

The Global Environment in the Knowledge Revolution*

Graciela Chichilnisky
UNESCO Professor of Mathematics and Economics,
Professor of Statistics and director of the Program on Information and Resources, Columbia University, 628 Mathematics, Columbia University, New York, N.Y. 10027, email:gc9@columbia.edu.

1. Summary

Markets are a dominant force in the world economy, in many ways a force for change and progress. During the course of this century market economies have led the race for industrialization, overcoming planned economies and traditional agricultural societies. The rapid expansion of international trade during the second part of the 20th century[1] has led to an extensive extraction of resources in developing nations, resources that are exported and consumed mostly in industrial nations and traded often at prices which are below replacement costs. This pattern of North-South trade in resources underlies the main global environmental problems we face today: biodiversity destruction and the risk of climate change. As the century turns, the market itself is evolving. Two major trends, the knowledge revolution and global environmental stress, lead to new and different types of markets involving a combination of private and public features and to new global challenges[2]. Environmental markets and markets for knowledge-based products[3] will play a pivotal role in the new century. Together they are becoming a critical

* "Knowledge revolution" is a registered trademark of the author. This paper was presented at the conference "Managing Human-Dominated Ecosystems" organized on March 27-29 1998 by a committee chaired by the author and including also Geoffrey Heal, Paul Ehrlich and Gretchen Daily, at the Missouri Botanical Garden. The valuable support of this committee and of Peter Raven, Director of the MBG, is gratefully acknowledged, as are the valuable comments of the participants and the staff at the MBG.

[1] Since World War II world trade increased at least three times more than world production, leading to the globalization of world trade. Even the USA, traditionally an isolated country, has more than doubled the proportion of trade to economic activity so that trade today accounts for 30% of US GNP.

[2] The agenda of the Program on Information and Resources (PIR) at Columbia University is addressed to these two major trends. This research was supported by the Sloan Foundation.

[3] Such as software or biotechnology products.

source of wealth and dynamics of the world economy, and could lead to the creation of a sustainable society. To realize the enormous potential of these new markets, new institutions are needed. This article proposes such new institutions, basing its analysis on a somewhat unexpected connection between intellectual property rights and appropriate property rules regarding the environment. Leading ecologists regard the genetical information encoded in biodiversity as representing the most important value of biodiversity, a form of intellectual property. This paper proposes a new regime for property rights that can encourage both an optimal production of knowledge and at the same time a sustainable use of environmental resources, to help achieve a wide diffusion of the benefits from both.

The vision ahead of us is that of a global economy which is very innovative in the use of knowledge but conservative in the use of resources, a society that is centered on diversity and human capital and offering the prospect of substantial economic progress without damaging the ecosystems that support life on earth.

2. The New Global Markets

Markets are a dominant institution in the global economy. As the century turns, however, the market itself is evolving. Two major trends are environmental markets and markets for knowledge. Environmental markets are starting to emerge as a response to global environmental concerns. The Chicago Board of Trade started trading emissions of sulphur dioxide (SO_2) following the US Clean Air Act and water markets are contemplated in California. The first global environmental market is being created: following our earlier proposal to the UN Framework Convention on Climate Change[4] the 166 nations who are parties to the Framework Convention for Climate Change (FCCC) agreed in Kyoto, December 1997, to create an international framework to trade carbon emission credits among industrial nations.[5] Related proposals were advanced in Barrett (1991 and 1992), Chichilnisky and Heal OECD (1994).

[4] Chichilnisky (1995a and 1995b).

[5] The countries involved in Article 17 of the Kyoto Protocol on emissions trading are mostly industrial nations. Our proposal differs from others in that it points out that differentiated property rights allocations are needed in emissions markets in order to reach efficient solutions in emissions trading. All other proposals consider the allocations of initial endowments as a matter of justice or ethical concerns for distribution. I advanced the proposal for the creation of an international framework for trading emissions permits at an international OECD conference in Paris, in 1993, and in 1994 at a workshop of "Joint Implementation and Beyond" organized under the auspices of the Global Environment Facility (GEF) with the participation of the members of the Bureau of the International Negotiating Committee of the Framework Convention on Climate Change (FCCC) at Columbia Business School in May 1994. In December 1995 the proposal for the creation of an International Bank for Environmental Settlements (IBES) that would organize and regulate emissions trading was presented officially at a keynote address to the Annual Meetings of the World Bank, Washington, D.C., and in various publications proposing blueprints for this trading regime, see Chichilnisky (1996a, b). In November 1997 The Rockefeller Foundation and the Global Environment Facility organized a workshop to discuss the creation of the IBES in Bellagio, Italy. In December 1997, Article 6 of the Kyoto Protocol, paragraphs 1 and 5, formalized the creation of such an international framework. The actual modalities, regulation and monitoring of the trading of emissions was discussed further at the Conference of the Parties (COP4) of the FCCC, which took place in Buenos Aires, November 1998. The PIR at the Columbia Earth Institute organized a follow-up conference for the UN FCCC in April 1999.

Markets for knowledge hold the key to the dynamics of the world economy: telecommunications and electronics, biotechnology and financial products, all involve trading products that use knowledge rather than resources as the most important input.

Markets for knowledge and environmental markets are unusual because they trade a different type of good, which I call "privately produced public goods". These are different from the private goods that characterize traditional markets. With private goods—such as apples or machines—traders can choose what they wish to consume independently of each other. Knowledge and environmental goods are different: the planet's atmosphere is the same for all, and knowledge can be shared without losing it. As explained below, knowledge and environmental assets are not private goods but rather privately produced public goods. Markets trading such goods will be important in the future, because knowledge and environmental resources are key trends in the world economy. These trends lead the transformation that I call the "knowledge revolution"®.

Focusing on these new markets, I analyze here the introduction of new institutions and the policies that can lead the transformation of industrial society into a sustainable society. I focus on a new type of economic organization, involving markets that trade a mixture of private and public goods. These new markets require new regimes of property rights, also proposed here. They carry with them the seed of a society which encourages the creation of knowledge, and could lead to a better use and distribution of knowledge and of the world's natural resources.

3. The Global Environment

The rapid pace of industrialization since World War II contributed to biodiversity depletion and to an unprecedented level of greenhouse gas emissions, both of which have become global concerns. Industrialization is based on energy. All over the world energy has been, and continues to be, based on the burning of fossil fuels, and the attendant emission of carbon dioxide. Scientists now know that carbon and other greenhouse gas emissions can cause climate change. After many years of searching for the causes, a consensus is emerging that industrial activity is the fundamental driving force of human-induced climate change. The success of international markets in propelling industrialization has magnified the use of fossil fuels and other natural resources worldwide. By propelling industrial society forward, global markets have fostered an excessive use of natural resources and the attendant emissions.

The enormous growth in the international market plays a pivotal role in the environmental agenda because this market mediates the relationship between industrial and developing countries, the North and the South. The developing South specializes in resources, which account for 70% of Latin American exports and almost entirely for those of Africa, while the industrial North specializes in products intensive in capital and knowledge. With few exceptions, economic development can be read from the composition of a country's exports. The most successful industrializing nations, the Asian Tigers, have swiftly moved into technology-intensive products,

and have shaped their markets to fit their development needs. Those nations that have been left behind, mostly in Africa and Latin America, have followed resource-intensive patterns of development. The handwriting on the wall is clear: resource-intensive development is not reliable and generally does not work. It should be replaced by knowledge-intensive development. In other words: The knowledge revolution may be the only way into a sustainable future.

The origins of the global environmental dilemmas we face today, while complex, are not difficult to trace. They involve the historical coupling of two different worlds: the industrialized and the developing regions, the North and the South. Since the end of colonialism, international markets have perpetuated a pattern of economic development in which, as already pointed out, the world's less advanced countries play the role of resource producers and exporters. What is not generally known is that this pattern of trade is explained in great measure by the historical difference in property rights between the North and the South. It is well documented that countries in the latter hold most resources as common property; in industrial economies these are on the whole private property. Recently differences in property rights have been invoked successfully in explaining the fact that the South over-extracts natural resources for the international market, selling these below real costs.[6] As a result, the North overconsumes resources and the South overextracts them. In a world where agricultural societies trade with industrial societies, markets magnify the extraction and as a result exports of natural resources exceed what is optimal[6].

4. Ecology and Knowledge

A major challenge today is to find practical paths for sustainable development. This means finding ways to reorient consumption patterns and use of natural resources in ways that improve the quality of human life, while living within the carrying capacity of supporting ecosystems.[7] This requires building a future in which humans live in harmony with nature. We are far from this goal. In many ways the world economy is moving in the opposite direction.

Just as the environmental problems generated by industrial society are becoming a threat to human welfare, however, industrial society started a process of transformation. The rapid pace of this change has led me to call it a revolution. The change is centered in the use of knowledge and for this reason I call it the "knowledge revolution." What characterizes this so-called knowledge revolution?

The question is best answered in a historical context, by contrasting the current situation with the agricultural and the industrial revolutions, two landmarks in so-

[6] See Chichilnisky (1994a, b). An "agricultural society" is a society that has gone through the agricultural revolution but has not completed the stage of "industrial revolution," so most output is raw materials or agriculture. In the "industrial society" most output is industrial manufactures.

[7] This is the definition of sustainability adopted by the Bruntland Report (1987), and is anchored in the concept of development based on the satisfaction of "basic needs," a concept that was introduced and developed empirically in Chichilnisky (1997a and b). Sustainable development is explored also in Caring for the Earth, a joint publication of IUCN, UNEP and WWF.

cial evolution. Neither of the two previous revolutions (agricultural and industrial) is complete. Across the world we find today pre-agricultural societies populated by nomadic hunters and gatherers, and most of the developing world is still within agrarian societies. While the two previous revolutions are still working their way through human societies, knowledge is becoming a leading indicator of a third wave of change. Knowledge means the ability to choose wisely what to do, and how to do it. This ability is becoming the most important input of production, and the most important determinant of wealth and economic progress. The key input in this revolution, knowledge, resides mostly in human brains rather than in physical entities such as machines or land. It is worth pointing out that the important input is *knowledge* rather than *information*. The difference between these two terms is the difference between the computer industry, which is based on information technology, and other sectors such as telecommunication, biotechnology and financial sectors, which involve knowledge rather than information technology itself. Information facilitates knowledge, and inexpensive information fosters new knowledge. It has been observed that the key value of biodiversity resides on its knowledge content.[8]

It can be said that knowledge is the content and information is the medium. Knowledge is driving change, and this is facilitated by the medium, information. Information technology is like a fuel for knowledge. Its abundance and inexpensive supply fuels the growth of sectors such as communications, biotechnology and global finance. Information technology fuels knowledge sectors because it performs the important role of allowing the human brain to expand its limits in the production, organization and communication of knowledge. The most important input of production today is not information technology itself: it is knowledge.

5. Characterizing the Knowledge Revolution

We may characterize the knowledge revolution as a period of rapid transition at the end of which knowledge itself becomes the most important input of production, the most important factor driving economic progress and wealth. For example, today the knowledge content of biodiversity can improve public health and human welfare, and is identified as a crucial source of economic value. By contrast, in prior revolutions the most important inputs were land (in the agricultural revolution) and machines (in the industrial revolution).[9] Knowledge differs fundamentally from land and machines in that it is not "rival" in consumption. More on this in Section 7 below.

The knowledge revolution is already with us. Indicators include the fact that the value of corporations in the stock exchanges of the world is increasingly measured from their knowledge assets, such as discoveries, patents, brand names and innovative products, rather than from their capital base or physical assets. This means that knowledge-type assets (such as patents) are increasingly regarded as

[8] According to leading ecologists such as E. Wilson and T. Lovejoy; presentations to the National Academy of Sciences.

[9] Both land and machines became better utilized because of new knowledge, but the main inputs of production were then land and machines; now it is knowledge itself.

an important source of value of a corporation. At the level of the economy as a whole, knowledge of mathematics and sciences has become a good predictor of national economic progress across the world; an illustration is in Figure 1 below.[10]

In this period of change the USA leads the pack. Today more Americans make semiconductors than construction machinery. The telecommunications industry in North America (USA and Canada) employs more people than the auto and the auto parts industries combined. The US health and medical industry alone have become larger than defense, and also larger than oil refining, aircraft, auto, auto parts, logging, steel and shipping put together. More Americans work in biotechnology than in the entire machine tools industry. Most US jobs in the last twenty years were generated in smaller, knowledge-intensive firms driven by risk capital. In the US, one third of the nation's growth is accounted for by the knowledge sectors, see Figure 1 below,[11] so that knowledge is an increasingly important determinant of economic progress. The knowledge sectors of the US economy already grow several times faster than the rest of the economy, and account for much of the dynamics of its economic growth. Increases in personal spending in the US show the trend clearly. For example, key old economy items such as motor vehicles, major food appliances and clothing grew on average 2.3% in 1997-8, while key new economy items such as home telephone services, entertainment and recreation services, cable TV, brokerage and other financial services and home computers grew on average 12.5%.[12]

Knowledge sectors consume less resources and have less ecological impact than the rest; thus they could decrease environmental damage once they become dominant in the economy; see the figures below. The question is whether the pace and scope of this process of change will foster a sustainable society in a time scale that matters. Encouraging and accelerating this transition is key. The economic transformation depends among other things on the evolution of the new markets for knowledge and for environmental assets. These require special analysis since, as already mentioned, knowledge and environmental assets are privately produced public goods, leading to new types of markets with new challenges and new opportunities for action.

6. A Service Economy?

It is important to differentiate the knowledge revolution from a service economy which used to be thought of as the latest stage of the industrial society. A service

[10] Data from TIMSS: Third Mathematical and Science Study, American Federation of Teachers, American Department of Education.

[11] See also *Business Week*, "The New Economy: What it really means" by Stephen Shepard, Editor-in-Chief, November 17, 1997, p. 40, last paragraph.

[12] See also Business Week, March 23, 1998. This is despite the fact that current systems of accounting undervalue the contribution of electronics, which are extraordinarily productive and offer rapidly lowering costs for their products, so their weighting factor in GDP (market prices) decreases with time. In a nutshell: in the US knowledge products are rapidly becoming the most important input of production, source of value and economic progress. Similar statistics hold in many of the OECD nations. Development of knowledge sectors is slower in Europe than in the US because their financial markets and property rights systems are not as flexible and as well developed and regulated. This is discussed further below.

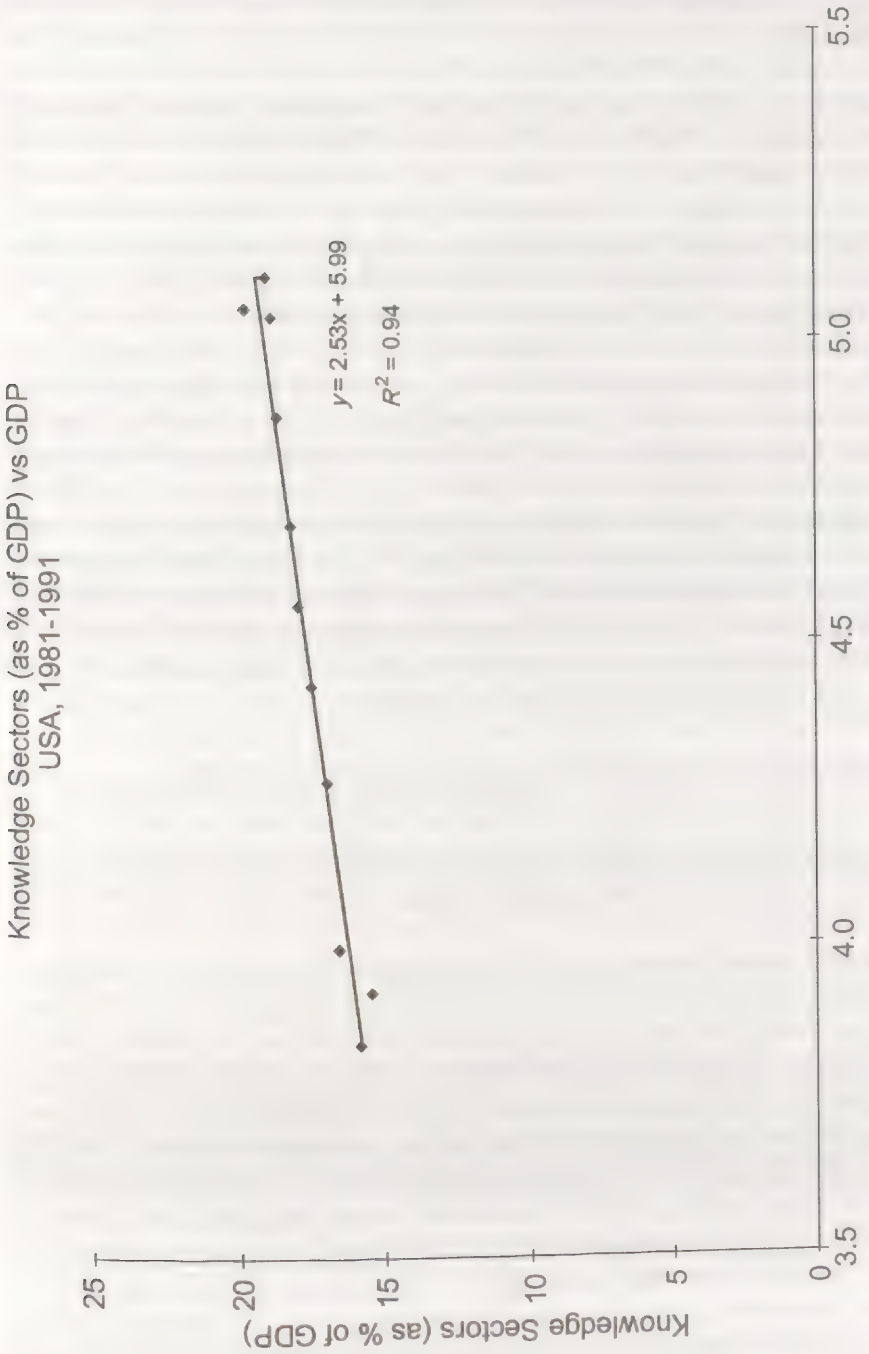

Knowledge Sectors (as % of GDP) vs GDP
USA, 1981-1991
y = 2.53x + 5.99
R² = 0.94
Knowledge Sectors (as % of GDP)
GDP (In trillion constant 1987 US$)
25
20
15
10
5
0
3.5
4.0
4.5
5.0
5.5

Figure 1.

economy is characterized by the production of services more than goods, and it is similar to a knowledge economy in that knowledge sectors often involve services (such as finance). It is true that services now make up the largest part of advanced industrial economies. However the analogy ends there. An inevitable concern about the service economy is that it could lead mostly to service-oriented labor, such as the labor employed in the food services or in bank processing, requiring little skill and therefore leading to lower wages. The main difference between the service economy and the knowledge society is that in the latter the typical worker is highly skilled and generally well paid. Furthermore the worker's knowledge resides in her/himself and her/his brain and life experience, rather than in the machines that complement labor. Therefore the knowledge economy could result, with proper institutions, in a society that is more human oriented than the industrial or the service society.

A distinct possibility is that in the next century a new society will develop, one that is centered in human creativity and diversity, and which uses information technology rather than fossil fuels to power economic growth. The vision is a human-centered society which is deeply innovative in terms of knowledge and at the same time very conservative in the use of natural resources. The patterns of consumption and resource may not be as voracious as those in the industrial society, and may be better distributed across each society and across the globe. The knowledge society may achieve economic progress that is harmonious with nature. This vision is only a possibility at present. Without developing the right institutions and incentives this possibility may never come to pass, and a historical opportunity may be lost. We need institutions to bridge the gap between a grim present and a bright and positive future. The rest of this paper will address this issue, for which an economic analysis of knowledge and environmental markets is required.

7. Knowledge and Resources as Privately Produced Public Goods

A key connection between knowledge and the environment is that both are public goods that are produced privately. This unusual connection between "public" and "private" aspects can be fruitful in understanding the new institutions needed for a global economy where knowledge and environmental markets acquire increasing importance.

First observe that environmental markets often trade *public goods*, by which we do not mean goods available publicly but rather goods that are not "rival" in consumption. An example is the concentration of carbon dioxide in the planet's atmosphere. Carbon mixes uniformly across the entire world, and therefore its concentration is one and the same for all. In this sense the carbon concentration in the atmosphere is not 'rival' in consumption. The second distinguishing characteristic of environmental markets is that the public goods traded are not standard, since they are generally *privately produced*. By contrast, classic public goods, such as law and order, are produced in a public, centralized fashion: by governments. Many environmental goods are produced by individuals in the course of their everyday

life: for example, through driving cars and choosing to heat our homes we "produce" carbon emissions and therefore influence atmospheric quality. In brief: environmental markets involve *privately produced public goods*.

The parallel with knowledge is now immediate. As knowledge itself becomes the most important input to production in the knowledge revolution, the economy changes because knowledge is also a special type of good. It is called a *public good* by economists, because knowledge can be shared without losing it and is therefore not "rival" in consumption. This is a *physical* property of knowledge, not an economic property, and as such it is quite independent from the organization of society. Nevertheles, the economic rules governing the use of knowledge—for example whether patents can be used to restrict its use—can have a major impact on human welfare and organization; more on this below. Knowledge is also different from conventional public goods of the type that economists have studied for many years, such as law and order or defense, which as already pointed out are typically supplied by governments, in a centralized fashion.[13] While knowledge is a public good at the level of consumption, it is produced mostly by *private* individuals. At the level of production, therefore, knowledge is like any other private good: costly to produce, and the resources used to produce knowledge often cannot be used for other purposes. Producing knowledge requires economic incentives similar to those for producing any other private good.[14] Hence knowledge is a *privately produced public good*,[15] exactly as environmental goods often are.

8. Markets for Privately Produced Public Goods

If current trends persist and markets continue to be a dominant institution in the global economy, we can anticipate that markets for knowledge and environmental markets will acquire increasing importance. However both of these markets trade privately produced private goods and are quite different from standard markets.[16] Such markets require special institutional arrangements:[17] they are efficient under conditions which

[13] Classic work in the area of public goods by Lindahl, Bowen and Samuelson, as well as modern work on the subject, analyze public goods as government policy, rather than in the context of competitive markets; see e.g. Laffont (1977).

[14] It is important to observe that this refers to a purely physical characteristic that is independent of social arrangements, and certainly does not contradict the obvious fact that it is possible to obtain economic gain by monopolizing knowledge within a given institutional environment. It is equally possible to monopolize any public good (air, water) and obtain economic benefits from this if allowed by the social institutions. For example, a bandit can set up by the New York water source and collect for himself a fee from all of those who wish to obtain it until the law catches up with him or her. Patents allow some of this behavior for a limited period of time, which is why they are considered inefficient.

[15] Knowledge means knowing what to do and how to do it. I use here a technical definition of a 'public good,' which does not imply that it is offered by the government. Indeed, in today's societies, knowledge is mostly produced and traded individually.

[16] The study of markets with public goods was formalized later by Foley in a general equilibrium context. The foundations of the modern theory of privately produced public goods can be found in the excellent work of Laffont (1977).

[17] Several proposals have been made in this direction, prominently Lindahl's theory of markets in which different traders pay different prices, a theory which is theoretically flawless but usually fails on practical grounds.

are much more stringent than those under which ensure efficiency in standard markets;[18] see Laffont (1977). Unless they are properly regulated, markets with public goods will lead to inefficient allocation of resources. Economists have shown this to occur because there are "externalities" that are not properly "internalized". Such externalities drive a wedge between the costs from the benefits. Aligning costs and benefits is an essential feature in the successful performance of the market as an "invisible hand". For example, in the case of markets for emissions permits, each trader has an impact on everyone's welfare through their emissions, yet their private actions in choosing how much oil to burn do not take into account the benefits that their emission abatement could produce for others. This miscalculation leads the economy as a whole to underinvest in the public good. In our case, this leads to carbon emissions that exceed what is efficient or optimal. This may well represent today's problems of global atmospheric quality: each country benefits the entire world when abating their carbon emissions. Yet the benefits they receive are only a fraction of the total, thus leading to less abatement than would be optimal for the world[19]. The key to resolving this problem is to find a way to "realign" costs and benefits of environmental decisions as it is the case in private markets, so that those who suffer the costs are in a position to take the actions needed to achieve the corresponding benefits.

To solve the dilemma of inefficiency in markets with public goods, Lindahl suggested using an ingenuous institutional framework involving "personalized" prices, a solution that is theoretically correct but is widely regarded as unrealistic.[20] Another solution involves the government itself determining the quantity of the public good produced, rather than market forces.[21] However this solution does not work when the public goods are *privately produced*. For example, emissions are produced by individuals in the course of their private lives, in burning fossil fuels for transportation or for home heating. It is not reasonable to expect goverments to control how much people drive their cars, or how and how much they heat and cool their homes. Similarly, knowledge production cannot easily be controlled by government action. Therefore government determination of the allocation of privately produced public goods is also impractical, like personalized prices are. New solution are needed today.

9. Property Rights Foster Market Efficiency

This article proposes an alternative way of recovering efficiency in economies with privately produced public goods, one that has not been considered until now: a two-

[18] For private goods.

[19] See Heal (1999) for more details.

[20] In this scheme, different traders pay different prices, depending on how they value the environment in relation to private goods. He showed that when using such prices, markets reach efficient solutions. Lindahl's solution is considered unrealistic, because one trader can "buy" from another the right to pay less, therefore inducing arbitrage and defeating the purpose of charging according to the trader's values. Is scheme could lead to 'black market' operations, and to a totally different outcome from that intended, an outcome that is no longer efficient.

[21] The optimality conditions were also determined by Lindahl, Bowen and Samuelson.

part system involving (1) the creation of *competitive* markets for trading rights in the use of the public goods[22] coupled with (2) specific allocations of initial property rights in the use of the public goods. We show below that, when taken together, these two steps can achieve Pareto efficient allocations of resources in markets involving both private and public goods. Both steps are needed to achieve efficient solutions.

The first step proposed above, (1), is relatively well known and understood today for emissions. Examples are emissions markets where traders are assigned a quota on their emissions, which they can trade freely among themselves. In the case of knowledge, however, the examples are less well known. I propose below the creation of markets for compulsory licences in the use of knowledge such as genetical information encoded in biodiversity. Such licences are substantially different from patents, as explained below, and avoid some of their shortcomings. The traders are assigned initial quotas in the use of knowledge (e.g. a given number of licences) which can be traded at competitive market prices. Both of these examples, emissions markets and markets for licences, illustrate part (1) of the proposed policy, and are relatively straightforward.

Part (2) of our proposal is much less understood. The use of specific initial allocations for ensuring efficiency in markets with public goods is new. The fact that one can reach efficient solutions by specific allocations of property rights (called here "efficient" property rights) is a somewhat unusual observation. It is rigorously based on general equilibrium results which are reported in the Appendix. In the case of emissions markets the new results of Chichilnisky, Heal and Starrett (1993) and Chichilnisky (1993a) show that *some but not all* allocation of initial property rights lead to efficient allocation of resources in emissions markets. This is true even though these are competitive markets. In the Appendix we show that the same result holds in knowledge markets involving markets for compulsory licences which have a structure that is similar to emissions markets.

As already mentioned, the specific allocations of property rights that lead to efficient market solutions are called **efficient property rights**. For the reasons explained above, these property rights regimes are particularly important both in markets with knowledge and in environmental markets, both of which involve privately produced public goods. The knowledge revolution, with additional institutions, can solve global environmental stress. Hence the title of this paper.

The use of property rights regimes to induce efficient solutions is a realistic policy because many property rights on environmental use are still to be determined. Indeed, through the Kyoto Protocol, emission rights on a global scale are in the process of emerging. So are property rights regimes on knowledge or on biodiversity through the United Nations Biodiversity Convention. One hopes that there is still time to develop policies involving efficient regimes of property rights for the environment and knowledge.

[22] In a competitive market all traders face the same prices, so that what is proposed here eliminates Lindahl's markets with personalized prices. Government action in terms of choosing the total level of the public good, as proposed by Lindahl, Bowen and Samuelson, is also eliminated by our condition of competitive markets. Our solution differs therefore from previous ones.

What allocations of initial rights are likely to lead to efficient outcomes? A rule of thumb is that those who own fewer private goods should be assigned proportionately more rights to use the public goods. In other words: as shown in the appendix to this paper and in Chichilnisky (1993a) and Chichilnisky and Heal (1994) market efficiency often requires that the allocations of initial rights to use knowledge or resources should benefit proportionally more those who have relatively lower income. This is a novel result in economic terms, even though it has an intuitive appeal at the policy level. For example, most school systems treat preferentially lower income groups, which often have free or subsidized access to education or to professional training schemes. But this is usually done on equity grounds, not on efficiency grounds. What is unusual here is that we offer a rigorous explanation of why choosing initial allocations that favor lower income groups is important for market efficiency. This is a matter of efficiency, not a matter of ethics. As industrial economies evolve into knowledge economies, access to knowledge becomes essential in determining wealth and welfare.

The application of these results for the case of biodiversity is of special interest. By analogy with the emissions trading system, efficiency suggests that patents or genetical information be replaced by a system of compulsory, negotiable licences that are traded within competitive markets, and with initial rights allocated preferentially to lower income regions or countries. More on this below.

10. Property Rights Regimes for Environmental Markets

What are 'environmental markets' and how do they work? Environmental markets work by creating and allocating property rights on the use of environmental resources, local or global, and allowing them to be traded.[23] They encourage conservation by increasing the cost associated with resources, thus discouraging their use. For example, carbon dioxide markets assign a price to the right to emit the gas into the planet's atmosphere. Therefore they add a cost to the use of the atmosphere to emit greenhouse gases: This can be the cost of purchasing permits when one exceeds one's allotment. Or it can be the opportunity cost of using one's allotment rather than selling it at the market price. In all cases, environmental markets make resources more expensive and discourage their use. They can therefore induce more rational use of resources globally.

Environmental markets come in many forms. One can trade rights to the use of water bodies such as aquifers, or to the use of the atmosphere of the planet for disposing of greenhouse gases. Traders can be individuals or corporations. They can also be countries. Environmental markets already exist in the US: permits to emit sulphur dioxide are traded on the Chicago Board of Trade. Following the US Clean Air Act, electric utilities were assigned rights to emit SO_2 up to an overall

[23] This is an idea in the tradition of Coase: one of the earliest developments in environmental markets is Dales (1930), more recently e,g. Barrett (1992), Chichilnisky (1993a) and Chichilnisky and Heal (1994, 1995).

level. They were also given the ability to write contracts to trade these rights in open markets.

Environmental markets are a growing trend. In the Kyoto Protocol in December 1997, 166 nations endorsed the global trading of emission reduction certificates by industrial nations,[24] the first step in the creation of a global market for emissions trading. Article 17 of the Kyoto Protocol provides a new scheme, which is still in an embryonic form, following a proposal that was advanced to the diplomats of the Climate Conventions by the scientists of Columbia's Program on Information and Resources to the United Nations Framework Convention on Climate Change in May 1992.[25] This extended to a global domain the trading on sulphur emissions that emerged in the USA after its Clean Air Act, and opened a new era in market economics. Our proposal to the United Nations Framework Convention on Climate Change differed from others in that we showed that special allocations of rights to emit by different countries would be needed to achieve efficient allocations in competitive emissions markets.[26] Until then it was assumed that a competitive emissions market would be automatically efficient, and all considerations of allocations of property rights were restricted to the realm of ethical concerns for justice and redistribution. In showing that property rights allocations are a precondition for efficient emissions markets, we showed that these markets behave very different from standard markets where equity and efficiency are divorced from each other.

Environmental markets differ from standard markets in that the "invisible hand" must reconcile two major concerns in today's society: equity among the traders and the efficient operation of markets. As shown below both equity and efficiency are closely intertwined in environmental markets because these markets trade goods that have a combination of private and public features, where each trader's behavior is linked to the whole. The issue supports an intuitive concern for equity, which has been apparent in the environmental movement since its beginnings. The appendix provides results showing that such intuition is not misplaced.

11. Property Rights Regimes for Markets with Knowledge

To produce new knowledge, its creators need incentives. Incentives could involve artificial 'scarcity', by restricting the use of the knowledge by others. Patents on new discoveries work in this fashion, by restricting others' use of knowledge. This creates inefficiencies because any restriction in the sharing of knowledge is suboptimal, because knowledge can be shared at no cost. So restrictions on the use of knowledge are inefficient after knowledge is created. However, without some re-

[24] Formally Annex B nations.

[25] See Chichilnisky (1993a) and (1995b.)

[26] Because of the connection between efficiency and equity in emissions markets, we proposed the creation of an International Bank for Environmental Settlements (IBES) in 1993 to the UN Framework Convention on Climate Change, and made a similar proposal at an invited keynote speech the Annual Meetings of the World Bank in 1995.

strictions there may be no incentive to create *new* knowledge. I call this the paradox of knowledge. This paradox is at the heart of the success of the knowledge society, of its ability to bring human development for many and not only wealth for a few.

New property rights regimes are needed to deal simultaneously with the need to share the use of knowledge for efficiency, while at the same time preserving private incentives for production. The appendix contains a technical summary of how this would work in practice within competitive markets.

I propose substituting patents by a system of *licences* which are allocated in a specific way that ensures optimal use of knowledge, and which are then traded, in a *competitive* fashion, along with all other goods in the economy. In this new scheme, the right to use knowledge is unrestricted (licencing is compulsory) and by law everyone has access to it; however, users must pay the creator each time they use this knowledge. Since the licences are traded in competitive markets, they ensure that the creators of knowledge are compensated for their labor in a way that reflects the demand for their products and therefore their usefulness for society. Prices are uniform and set by competitive markets. Since licences are compulsory, they make knowledge available to all. In this sense, this regime differs fundamentally from patents because, in principle, patents can be used restrict the use of knowledge.[27] No restriction in the use of knowledge is allowed in the system I propose. However, a key issue is the initial distribution, property rights, use and applicability of the licences, to which we now turn.

It is clear that a system of licences on knowledge products (e.g. operating systems for software, biological information, how-to-do-it systems) could in principle preserve or even worsen today's uneven distribution of wealth in the economy. This is because the knowledge economy has a built-in incentive for the creation of monopolies. Indeed, any knowledge-based corporation is a "natural monopoly," a technical term used to indicate that the cost of duplicating knowledge products (such as a software products) is very small, and therefore the larger the firm the lower are its costs. This is an extreme case of "increasing returns to scale" where larger firms have an advantage over their competitors, and therefore can prevent entry by newer and smaller competitors. Such natural monopolies are frequent in the knowledge society. How to avoid concentrating knowledge in the hands of very few?

The system of property rights proposed here takes into account these possibilities. It establishes how the initial distribution of property rights on licences is a crucial element in achieving efficient solutions. The solution is to achieve a distribution of property rights on licences that is negatively correlated with the property rights on private goods, and beyond this to ensure that markets for knowledge act competitively. The results in the appendix make this proposal rigorous within a standard model of a market economy.

[27] Patents *can* be negotiated and licenced, but they do not *have* to be. Owners of patents are legally entitled not to licence their use, effectively creating a "monopoly" during the period of the patent's life. Compulsory licences do not create monopolies.

How can such a system of property rights become accepted? This concern parallels that proceeding the introduction of laws to ensure fair trade, a matter on which natural monopolies have offered and continue to offer much resistance and which is eventually overcome by society as a whole.

There are substantial economic incentives for corporations to accept fair trading and the systems of property rights that we propose, although more business education is needed before widespread acceptance. For example, those producers that benefit in principle from increasing returns to scale could support a system of licences in which the lower income segments of the population are given proportionately more rights to use knowledge than the rest. Consider as an example the case of worker training schemes. Because knowledge is so important for the productivity of society as a whole, and produces positive "externalities" on all producers, there is an incentive to develop a skilled pool of workers. Corporations know that the knowledge owned by skilled workers is essential to the success of their knowledge industries.

In a proposition presented in the appendix, it is shown that for an efficient market solution, one that cannot be improved so as to make everyone better off, lower income traders (individuals or nations) should be assigned a larger endowment of property rights in the use of knowledge. This means a larger amount of licences to use knowledge must be assigned to lower income countries or groups. The scheme is new but realistic. Similar systems are already in place in industrial societies. An example is the auctioning of use of airwaves by the US Federal Government in Washington, D.C.: minorities and women are given substantial discounts when they participate in auctions for the purchasing of property rights on the airwaves. In certain cases this involves a 40% discount of the auction prices.

12. Licences: We Make It, We Take It Back

The system of property rights proposed here, while unique in its economic formulation, is reminiscent of a development that is already taking place in the US corporate world. It is connected with environmental issues that have a public good aspect, such as the disposal of materials involved in heavy industrial products, vehicles and electronic equipment. Leasing vehicles (a form of licences for use) and electronic equipment is now a thriving business that hardly existed twenty years ago. For example, one of the largest packaging companies in the world, Sonoco Products Co., started taking its used products off customer's hands after its CEO Charles Coker made a pledge in 1990: "we make it, we take it back." The policy has already been adopted by the car industry in Germany, where car manufacturers are responsible for disposing of the vehicles that the customers return at the end of their useful life, due to environmental concerns. Another example arises in the floor covering industry. Ray Anderson, CEO of Atlanta-based corporation Interface, the largest maker of commercial carpeting, has set up as a goal to create zero waste while making a healthy profit, and takes back the used products that it sells to recycle them. The mission of

their businesses, all these business people say, is to sell licences to use *services*, rather than the ownership of products. In other words: rather than selling TVs, selling licences to viewing services; rather than selling vehicles, selling rights to use transportation services, rather than selling carpets, selling rights to the comfort and visual services that carpets provide. Licencing has the advantage that the producers have an incentive to minimize waste and environmental damage—for example, the waste produced by wrapping or by defunct car bodies—as they will be responsible for it. These business people see licencing services as the way to the future, particularly when they are confronted with paying for the disposal of industrial waste.

Implicit in this new system of property rights is an idea that we share: *licencing the use of services* rather than *owning the products that deliver those services*. The products in the corporate examples just described share another common characteristic with our economic approach: they have some of the characteristics of public goods that produce negative environmental "externalities." Knowledge, as we saw, also produces externalities, although positive.

Knowledge, as we saw above, has much in common with environmental assets: it is a privately produced public good. Knowledge products have been licenced for many years, although this has been done in a case-by-case manner, without securing the competitiveness of the market for licences, and without securing the distribution of property rights that would ensure efficient outcomes. In this sense, the new developments in industry reported here move in the same direction as the system of property rights, involving licences, proposed in the appendix and discussed above. *These new systems of property rights can be thought of as a step forward and institutionalization and economic formalization of licencing and leasing systems that have recently emerged in advanced industrial economies.*

13. Impacts of Property Rights on Knowledge

Rules to govern the use of knowledge lead to threats and opportunities for human development, both directly and through the possible changes in the patterns of consumption of goods and services. They can determine the impact of human societies on the environment and on resource use, as well as determine inequalities across the world economy. The way we use and distribute knowledge casts a very long shadow on human societies.

A historical comparison helps to explain this process. In agricultural societies the way humans regulated the ownership of land, which was then the most important input to production, led to social systems such as feudalism. Ownership of land had therefore a major impact on human welfare and on economic progress. Similarly in industrial societies the way humans organize the use of capital, which is its most important input of production, leads to very different social systems such as socialism and capitalism. Indeed, these two systems are defined by the rules on ownership of capital. In socialism ownership is in the hand of the gov-

ernments or other public institutions, and in capitalistic systems capital is in private hands. Property rights on capital have mattered a great deal, and have led to global strife in most of this century. Since capital is the most important input of production in industrial society, it is clear that property rights on capital had an enormous impact on the organization of society, on economic progress and on people's welfare.

Similarly in the knowledge society the way humans organize the use of knowledge, which is the most important input to production, will determine human welfare and economic progress across the world. This means that human institutions that regulate the use of knowledge, such as property rights and markets for knowledge, will become increasingly important. However, as we saw knowledge is a different type of commodity than land or capital: it is a public good. Markets with public goods, and other economic institutions such as property rights on public goods, are still open to definition and require much economic analysis. Markets themselves will operate differently in the knowledge economy, because the nature of the goods traded is different. There will be new challenges and new opportunities.

14. The Ecological Impact of Knowledge-Intensive vs. Resource-Intensive Growth

It is critical to distinguish two patterns of economic growth, two extreme cases of which are a spectrum of possibilities: economic development that is *knowledge-intensive*, and that which is *resource-intensive*. The former simply means achieving more human welfare with less material input. The latter means achieving more production by means of more material use. These two categories were introduced in Chichilnisky (1995a, 1994b).

There are excellent historical examples of the two patterns of development, and of the differences they induce on economic growth. As already mentioned, East Asian nations fit the knowledge-intensive paradigm, while Latin American countries and those in Africa, fit well the pattern of resource-intensive growth. On the whole knowledge-intensive development strategies succeeded, while resource-intensive development patterns lost ground. Chichilnisky (1997) studied the historical patterns focusing on East Asian nations that are now called the Asian Tigers, including Japan, Korea and Taiwan, and later those called the Small Tigers, such as Singapore, the Philippines, Hong Kong and Malaysia. These focused on exports of technology-intensive products such as consumer electronics and technologically advanced vehicles, and overturned the traditional economic theory of "comparative advantages." In contrast with East Asian nations, Latin America and Africa followed a resource-intensive pattern of development and lost ground.

15. Scenarios of Development in the North and the South

As already mentioned, the most dynamic sectors in the world economy today are not resource-intensive; they are, rather, knowledge-intensive, such as software and hardware, biotechnology, communications and financial markets (Chichilnisky 1994b, 1995a). These sectors are relatively friendly to the environment. They use fewer resources and emit relatively little CO_2. Figure 2 below illustrates this in the US economy. Knowledge sectors are the high-growth sectors in most industrialized countries.

Some of the most dynamic developing countries are making a swift transition from traditional societies to knowledge-intensive societies. Mexico produces computer chips, India is rapidly becoming a large exporter of software, and Barbados has recently unveiled a plan to become an information society within a generation (Fidler 1995). These policies are a natural extension of the strategies adopted earlier by the Asian Tigers, Hong Kong, Republic of Korea, Singapore, and Taiwan (Province of China), who have achieved extraordinarily successful performance over the last twenty years by relying not on resource exports but rather on knowledge-intensive products such as consumer electronics. By contrast, Africa and Latin America emphasized resource exports and lost ground (Chichilnisky 1994b, 1995a, 1995-1996).

The lessons of history are clear: not to rely on resource exports as the foundation of economic development. Africa and Latin America must update their economic focus. Indeed, the whole world must shift away from resource-intensive economic processes and products. In so doing, fewer minerals and other environmental resources will be extracted, and their price will rise. This is as it should be because today's low resource prices are a symptom of overproduction and inevitably lead to overconsumption.

Not surprisingly, from an environmental perspective one arrives at the same conclusion: higher resource prices are needed to curtail consumption.

Producers will sell less, but at higher prices. This is not to say that all will gain in the process. If the world's demand for petroleum drops, most petroleum producers will lose unless they have diversified into other products that involve fewer resources and higher value. Most international oil companies are investigating this strategy. Examples are British Petroleum and Shell.

The point is that nations do not develop on the basis of resource exports, and at the end of the day development can make all better off. As the trend is inevitable, the sooner the transition to the Knowledge Revolution, the better.

The data and a conceptual understanding of how markets operate lead to the same conclusion. Economic development cannot mean, as in the industrial society, doing more with more. It means achieving more progress with fewer resources.

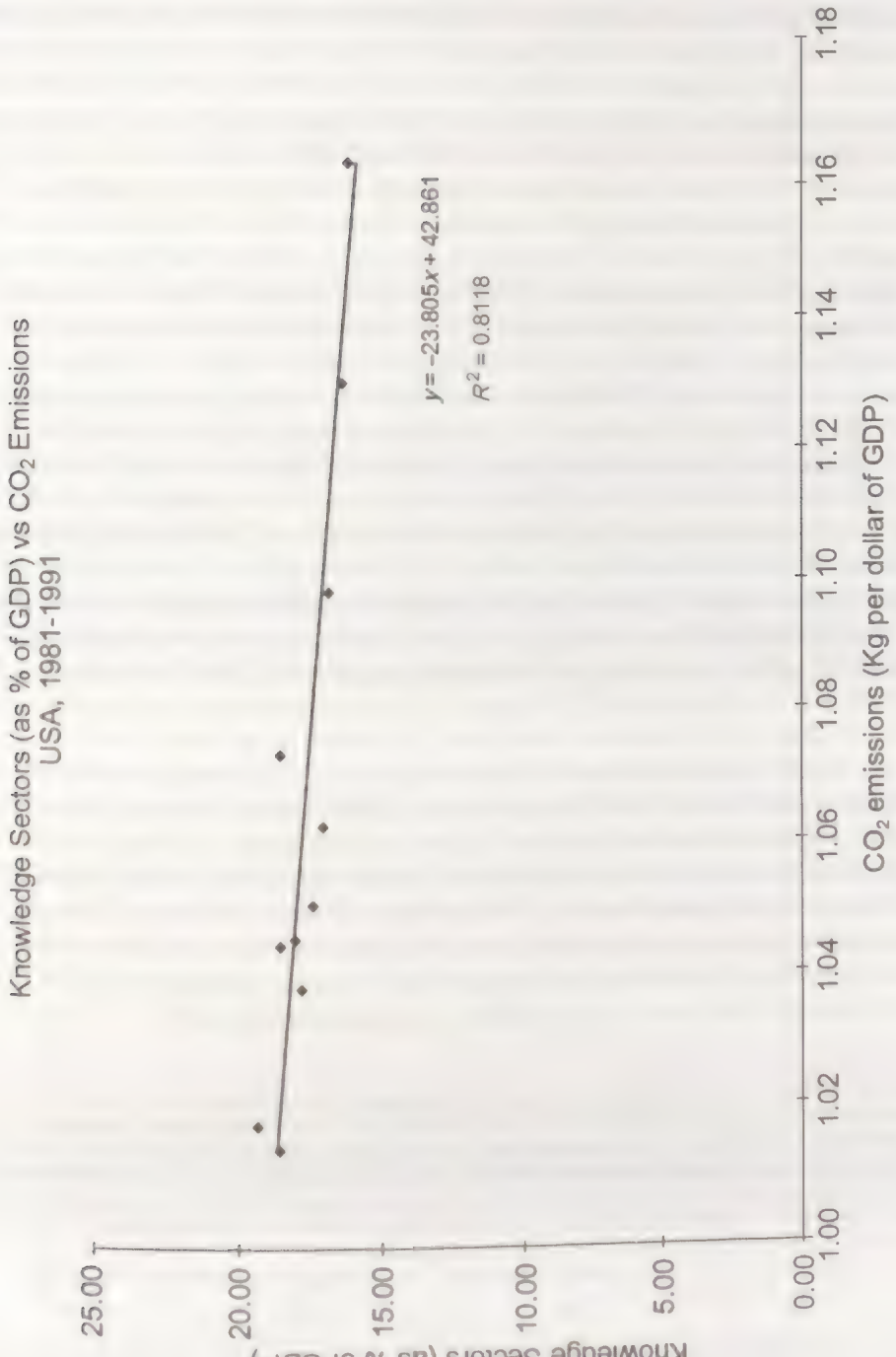

Knowledge Sectors (as % of GDP) vs CO_2 Emissions
USA, 1981-1991
$y = -23.805x + 42.861$
$R^2 = 0.8118$
Knowledge Sectors (as % of GDP)
25.00
20.00
15.00
10.00
5.00
0.00
1.00
1.02
1.04
1.06
1.08
1.10
1.12
1.14
1.16
1.18
CO_2 emissions (Kg per dollar of GDP)

Figure 2.

16. Policy Questions

(1) Should the World Trade Organization link environmental property rights to the trade regime? (2) Against the background of a free trade regime, is a movement in the direction of privatizing of resources always welfare improving? (3) What is the role of property rights on equity and efficiency? This section will offer answers based on the findings reported above and in the appendix.

It is well known that ill-defined property rights lead to resource overuse, and ultimately cause the destruction of the resource base itself. This has been called the "tragedy of the commons." Recently it has been shown that the commons problem is amplified by international markets (Chichilnisky 1994a, b). Since property rights on resources are lacking in the South, an expansion of resource trade may damage the environment and decrease welfare (Chichilnisky 1994a, b). In this sense, the pattern of North-South trade that we observe, which is largely responsible for the world's overuse of natural resources, originates from a historical situation involving the coupling of two types of nations through the international market: developing nations that have traditional institutions and common property rights on resources (the South) which are used as "open access" during the process of industrialization, and others with better defined private property rights on resources (the North).[28] Liberalizing international trade in this case amplifies the *tragedy of the commons* in the South: resources are overextracted and exported to the North at prices that are lower than optimal.[29] The North consumes more resources than is optimal.[30] The result is a misallocation of the world's resources, which are undervalued and overused. As already mentioned, this is a main source of the global environmental dilemma we face today. It explains the crisis in *biodiversity* destruction, and the problem of *climate change*. Both are based on the overuse of resources: forests in the first case, and minerals such as petroleum in the second. Both are overextracted in the South, exported at prices which are below real costs, and overconsumed in the North.[31] These results offer a positive response to questions (1) and (2) presented above. However, below we see that in certain cases the answer can be negative. One can say precisely when the answer is positive and when it is negative—more on this below.

[28] Although somewhat unconventional at the time, the Chichilnisky (1994a, b) results have become increasingly popular. They are based on economic theory and a historical fact: that industrialization proceeds by increasing privatization of resources. Natural resources such as forests, mineral reserves and biodiversity are often held as common property in the developing countries of the South, but as private property in the industrialized North. As a result resources in the South are used on a "first come first served" basis. The economic theory developed in my article explains that this leads to their overextraction and to a false impression of resource abundance and 'comparative advantage.' The reality can be quite different.

[29] Optimal prices are those that would prevail in the Pareto optimal solution: the equilibrium of a benchmark economy which is indentical to the one considered here but is perfectly competitive and has private property rights, see below.

[30] The definition of 'optimal consumption' parallels that of optimal prices: it is the consumption that would prevail in a general equilibrium model which is identical to that considered here but with no market failures.

[31] For data on this issue see *World Resources and the Environment* (World Resources Institute, 1994-5) and Chichilnisky (1995-6) "The economic value of the earth's resources."

This historical interpretation of the environmental problems we face today leads to another natural question: would it suffice to privatize natural resources in the South in order to resolve the overuse of environmental resources across the world? We identify below in which cases it does, and when it does not. In the former case the World Trade Organization should link environmental property rights to the trade regime and in the latter case it should not. This also gives an answer to questions (1) and (2) posed above. More precisely: privatizing in the South—namely the creation of private property rights on environmental assets—may not by itself solve the problem of overuse of natural resources in the world economy.

- In nations with common property regimes, international trade can lead to the *underuse* or to *overuse* of resources. The outcome depends on whether the common property regime affects inputs or outputs. Environmental assets are usually inputs, which is the reason liberalization does not work. In the case of outputs, for example software, the situation can reverse the outcome.
- Within free trade regimes, privatizing may or may not improve welfare. It can fail to do so when the goods privatized are environmental assets. Examples include the global market for carbon emissions created recently by the Kyoto Protocol.
- The distribution of property rights among the traders can be decisive in achieving efficient outcomes. Giving more rights to lower income groups can increase overall efficiency. This is relevant for the global negotiations in climate change.

These issues are analyzed in the appendix within a model of North South trade similar to one introduced in 1981(Chichilnisky 1981, 1994a). This differs from a standard trade model in that resources in the South are held as common property.

Below I draw also on historical evidence for these new questions, involving contemporary history. In Kyoto, December 1997, 166 nations agreed to assign ceilings on industrial countries' emissions of greenhouse gases, i.e. property rights on the use of the atmosphere,[32] and on a system of trading these property rights that I proposed in 1994 to these policy makers.[33]

17. Are Emissions Markets Efficient?

Consider a resource which is extracted from a *commonly owned pool* in the South: for example, petroleum residing on national soil. According to Chichilnisky (1994),

[32] The so-called Annex 1 countries.

[33] I.e. the Bureau of the International Negotiating Committee (INC) of the Secretariat of the Framework Convention of Climate Change (FCCC). As mentioned above, this proposal was presented at a seminar sponsored by the Global Environment Facility (GEF) and organized by the author and Professor G.M. Heal at Columbia University School of Business in May 1994. In December 1995 the proposal was officially presented at a keynote address to the Annual Meetings of the World Bank, and it was subsequently published under the auspices of UNESCO and UNDP in September 1995, see Chichilnisky (1995b).

this would lead to excessive extraction of petroleum in the South and to its specialization in petroleum even when it has no comparative advantage; to lower international prices for petroleum, and to overconsumption of petroleum in the North.[34] An example that comes to mind is Mexico trading with the US. Mexico exports petroleum to the US but is known to have rather shallow reserves that are expected to run out soon in the next century. The US imports most of its petroleum from Latin America (Mexico, Venezuela and Ecuador) and its consumers pay prices that are 2.5 times lower than those paid by their German or Japanese counterparts. Corresponding to this, the US consumes today about 25% of all petroleum produced in the world and is less efficient in the use of petroleum (about 40% less) than is Germany or Japan. At the same time the US generates 25% of all the carbon dioxide emitted in the world. Not surprisingly, it has been a reluctant party in the global climate negotiations to decrease carbon emissions.[35]

The following formalizes the introduction of property rights, and shows how this may not lead to Pareto efficient outcomes. In the example just described the price of petroleum extracted from a common property pool is too low when compared with a benchmark economy with private property rights on the resource pool. The common property problem may be difficult to correct because governments in the South own the soil in which the resource resides, and sovereign rights make it difficult to interfere. Alternatively, petroleum reserves may be shared by several landowners who own the land, a case covered by the US Hot Oil act. This case would require new legislation within the country, and there may be difficulties in interfering with sovereign legal institutions. In any case, because of common property rights there is an excessive extraction of oil. Corresponding to this, there are excessive emissions of carbon dioxide, which are a byproduct of burning fossil fuels.[36]

While property rights on the soil may be difficult to change, it is possible to address directly the problem of excessive emissions of carbon dioxide. This can be achieved by an international agreement fixing a total ceiling on emissions, and giving to each country or region a right to emit that adds up to this total. Regions can trade their rights among themselves. Such a procedure amounts to assigning private property rights on the use of the atmosphere to the regions, and allowing them to trade.

This procedure could resolve the problems emerging from the common property of oil, by assigning property rights on the emissions of carbon which are associated with burning oil. The result would be the creation of an international market on carbon emission rights. This is what in contemplated in the Kyoto Protocol: 166 nations have agreed to this solution for Annex I countries in December 1997.[37] The question is whether this solution leads always to efficient allocation of resources.

[34] All compared to the benchmark case that is, as discussed above, taken as the 'optimal.'

[35] Annex I nations, see Article 6 of the Kyoto Protocol.

[36] The term 'excessive' means, as before, exceeding what is optimal under an optimal solution corresponding to a private property rights regime.

[37] See Article 6 of the 1997 Kyoto Protocol.

The answer is generally negative: the solution is typically not Pareto efficient, although there are cases where efficiency is obtained.

To understand why the solutions may not be efficient consider the simplest case: a North-South economy where each region produces one good y using one input of production x: energy (petroleum). The more energy is used, the more carbon dioxide CO_2 is emitted, i.e. the less is 'abated,' and the higher is the concentration of carbon dioxide in the atmosphere. Carbon dioxide diffuses very uniformly and stably, taking from 60 to 100 years to break down. As a result the whole world is exposed to the same atmospheric concentration of CO_2. Define the level of abatement or 'quality of the atmosphere' a as the inverse of the concentration of carbon dioxide: the more carbon is emitted, the worse is the atmosphere's quality.

Better atmospheric quality means a more stable climate and therefore more welfare, so the utility functions of the regions increase in a; they also increase with the level of consumption of goods, y. In this situation, atmospheric quality is a public good which is available to everyone in the same 'quantity,' and is therefore not 'rival' in consumption.[38] However a is not produced by the government as are classic public goods such as law and order. The quality of the atmosphere a is a *privately produced public good* because driving a car, or using energy are private activities. Every individual in the North-South economy affects the composition of the atmosphere, and in this sense 'produces' the public good a. For the purposes of this section, the most relevant aspect is that due to its physical characteristics[39] a is available to everyone in the same amount: individuals cannot choose different atmospheric qualities. This links closely the consumption of different individuals in a way that does not occur with standard private goods, and requires special allocations of property rights in order to reach efficient solutions.

In a nutshell: despite the privatizing of the global commons (the atmosphere) a competitive equilibrium of the North-South model may or may not satisfy the additional Lindahl-Bowen-Samuelson condition for efficiency which derives from the fact that a is a public good. Therefore:

- Despite the privatizing of atmospheric use, the competitive market equilibrium may not be Pareto efficient. This is due to the physical characteristics of the atmospheric concentration in CO_2, which is the same across the planet, thus making a a privately produced public good. Property rights policies can lead to efficient solutions.

To show how an equilibrium in these markets may be Pareto inefficient, it suffices to provide an example. In the following diagram 1 the first market equilibrium is Pareto inefficient: this is shown by exhibiting a second market equilibrium in which both regions, the North and the South, are both strictly better off. The horizontal

[38] This is a definition of a public good.

[39] CO_2 mixes very thoroughly and stably across the planet's atmosphere, so its concentration is the same the world over. Sulphor dioxide does not have this property.

axis denotes quantities of the private good y_i and the vertical axis measures abatement a_i and the quality of the atmosphere, a. Here $i = 1, 2$ denotes the region (North and South). There is an initial allocation of emissions rights as indicated in the vertical axis for the North and the South. The first equilibrium has permit prices denoted "initial prices"; at those prices each region maximizes profits by producing at the respective levels of initial production, and using energy corresponding to an abatement level denoted initial abatement. At the initial allocation the North has an excess of permits which it sells to buy additional private goods; region one exports permits to import private goods, and the opposite is true for region two. At an equilibrium both markets clear. This equilibrium is inefficient in the sense that both regions can be made better off within the existing ceiling on emissions, as shown in the diagram. The first situation depicts a situation similar to a prisoner's dilemma, in that both traders are in an inferior position. A second initial allocation of rights corresponds to a lower ceiling on emissions and leads to higher utility levels for both regions, thus moving the traders to a Pareto superior position. Note that in the second situation more rights to emit are given to the South and fewer to the North than in the first, and *the total level of emissions in the second equilibrium is within the limits of the first equilibrium, but the welfare of both traders is higher in the second equilibrium.* This example illustrates why privatizing may not lead to Pareto efficient solutions, as we wished to show.

In the diagram 1 provided below there are two ceilings on emissions; a second lower ceiling on emissions is allocated between the North and the South in a manner that favors the South, leading to increased efficiency for both North and South. This example illustrates the connection between the initial rights and the efficiency of the market.[40] This is in contrast with the standard markets with private goods, where *any* initial allocation of property rights leads to efficient outcomes.

The result is somewhat surprising in two ways, each pointing in the opposite direction from the other. For environmental economists, the result is surprising because markets with emission permits are generally thought to be efficient independently from the distribution of initial endowments.[41] This is true for certain types of emissions which can be treated as private goods, such as the particulates that tarnish laundry appearing in a well known example developed by R. Coase. But the physical characteristics of carbon emissions change the nature of the problem, since the atmospheric concentration of carbon, as discussed above, is a public good. In this case Coase's arguments do not work, and efficiency can be lost. This observation explains the difference between the results presented here and Coase's results, but makes the results surprising for another reason. The second reason is that markets with public goods seldom have Pareto efficient equilibria. This is a well known fact for

[40] A general result along these lines was obtained by Chichilnisky, Heal and Starrett (1993): they established that there exist a small set (measure zero) of allocations of property rights from which a market trading private goods as well as privately produced public goods achieves a Pareto efficient allocation.

[41] See for example Chapter 3 of Baumol and Oats's excellent book on the subject.

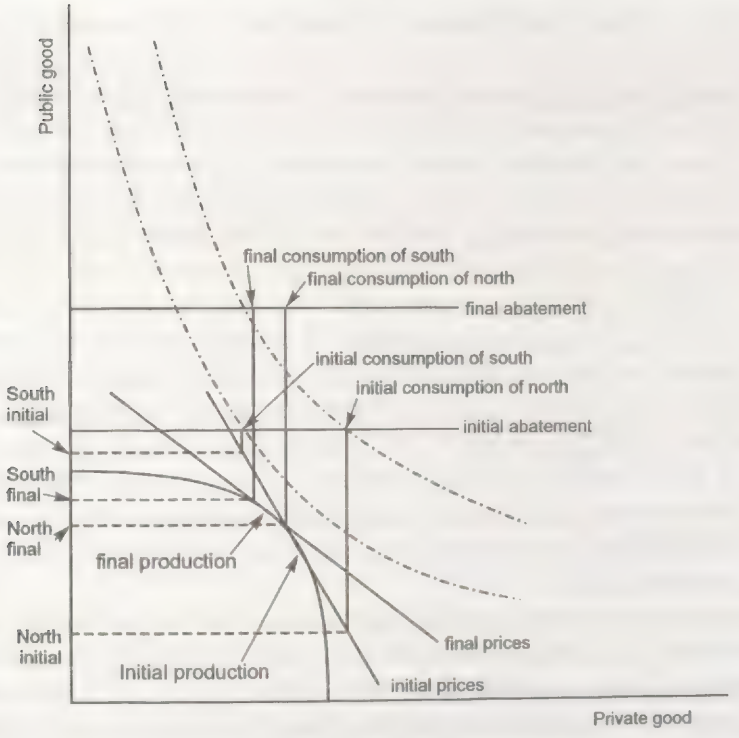


Diagram 1.

public economists. General results in this area until now are due to Lindahl and later Foley within a general equilibrium context, involving in both cases an extension of the market to include 'personalized' prices. Lindahl's solution requires that prices should reflect the characteristics of the traders, effectively multiplying the number of prices and equations in the market by the number of traders. As already mentioned, this allows 'arbitrage' by which traders 'sell' to each other the use of their personalized prices.

A classic result is that with personalized prices, markets with public goods always reach Pareto efficient allocations. The result obtained by Chichilnisky, Heal and Starrett (1993) suggests that rather than using personalized prices (and therefore introducing a large number of markets) one can reallocate property rights and obtain generically first best Pareto efficiency. Diagram 1 above illustrates how reallocating property rights among the two regions can improve efficiency and benefit both the North and the South.

18. Equity and Efficiency

We saw specific cases where privatizing does or does not lead to efficient outcomes. The free market solution may leave room for improvement for both the North and

the South. International coordination can therefore lead to superior outcomes for all trades.

The following diagram 2 takes a particularly simple case and develops a 'thought experiment' that helps to see why efficiency requires that those with lower endowments of private goods should generally be given a larger endowment of the public good—in this case, more rights to emit. The diagram is an Edgeworth box in which two traders trade two goods. In the standard case, every point in the box is an allocation of private property rights that, after trading takes place, leads to a Pareto efficient market equilibrium in the 'contract curve'. Now consider a new economy, where one of the goods is a public good, measured on the vertical axis. Now most of the efficient allocations on the contract curve cannot be realized in practice, because they involve traders using different amounts of the public good which is physically impossible. Therefore we must restrict our attention to those allocations within the contract curve in which both traders consume the same amount of a. Now observe that the initial endowments of rights that lead to such allocations allocate more public goods to those who own fewer private goods, as they are located on the budget line which is negatively sloped.

The inverse relation between property rights on private and public goods that was just discussed, and that is needed for efficiency, could play a useful role in the follow-up to the Kyoto negotiations.

Several countries have expressed the need for a policy of so-called "differentiation" involving the need to spread equally the burden of abatement. Australia advanced such a proposal in 1998; within the European Union such differentiation has already been incorporated: countries such as Portugal can increase their emissions by 40% while the December 1997 proposal for the entire EU was to decrease its overall emissions by 15%. This proposal to share the burden equally is consistent with the inverse connection between rights to emit and rights to private goods explained here. Our result is based on efficiency, e.g. the desire to achieve the agreed reductions on emissions with the minimum loss of welfare. The main point of this section is that these two issues, equity and efficiency, are closely connected in a market with privately produced public goods. This could lead to a valuable source of common interest between North and South as the negotiators assemble in The Hague in November 2000.

19. People-Centered Development: Opportunities and Threats

The knowledge revolution could develop in different ways, depending on the way our institutions and policies unfold. Knowledge has the capacity of amplifying current discrepancies in wealth, because it can lead to natural monopolies such as those that arise due to the adoption of operating systems[41] or other standards. In the North-South context, knowledge sectors could amplify the differences in wealth between the North and the South. If this occurs, then the low resource prices from developing countries will persist, since they are caused in part by the necessity to

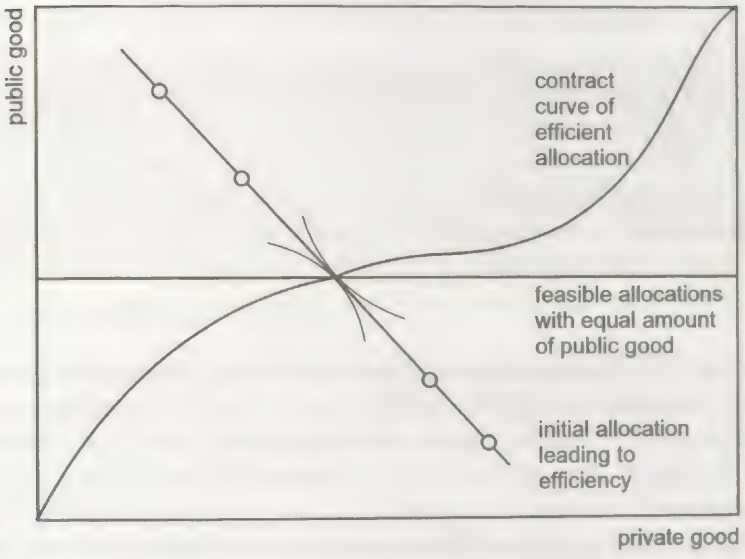


Diagram 2.

survive at low income levels within a difficult international market climate. It has been shown that with current institutions of property rights, anything that leads to more poverty leads to increased resource exports from developing countries (Chichilnisky 1994a).

On the other hand, knowledge sectors could flourish in those developing nations that have skilled labor. Several developing nations are, or could be soon, in that position. For example, the Caribbean and Southeast Asia are a case in point, as are many areas in Latin America (Harris 1994).

The main issues are to

- abandon the resource-intensive development patterns that these nations have followed for the last fifty years, with the support and encouragement of the Bretton Woods institutions such as the World Bank and the IMP, and
- seek to establish the institutions (property rights, financial structures) that could lead them to overcome the "comparative advantages" mirage, avoid the heavy stages of industrialization, and "leap frog" into the knowledge society.

Heavy accumulation of capital (financial or physical) is not needed for most knowledge sectors. What is needed is highly skilled labor, of the type that does not require expensive machinery or heavy capital investment in plants, and good managerial ability, all knowledge inputs that rely on a pool of abundant skilled labor. A good example is Bangalore, India's software industry, which produces about $2 billion of exports today.

20. Conclusions

Market-driven industrialization and the historical differences in property rights on resources in the North and the South are at the core of today's environmental problems. Almost paradoxically, markets could be part of the solution. In many cases, new markets for knowledge and environmental markets could help achieve a sustainable economy, and alleviate some of the worse imbalances in the distribution of global welfare, provided new regimes of property rights are adopted. This article suggests what forms these new property rights regimes could take.

Knitting together several issues into a coherent whole, this article has suggested responses to the following questions: Why has the global environment become such a concern today? What has led to a worsening of the environmental problem in the last fifty years? The response here is the tremendous success of the industrial revolution and its voracious appetite for energy. Are the developing nations, indeed the poor people of the world, central to a resolution of the problem? What are the connections between equity and efficiency in markets for knowledge and resources? The response here is that the overuse of natural resources is a North-South issue, deriving in part from differences in property rights over resources in the two regions. What is the role of international markets? Under today's conditions will the trend towards trade liberalization lead to a sustainable use of the world's resources? What policies are more likely to lead to a new form of industrialization that is innovative in the use of knowledge and conservative in the use of resources? What policies are more likely to lead to a new form of industrialization that is innovative in the use of knowledge and conservative in the use of resources? The response here are the new institutions such as licences for the use of knowledge and the environment, suggested here.

21. Appendix

21.1. The literature

Markets with privately produced public goods were studied by Laffont (1977) and others in a partial equilibrium world. This appendix looks at the problem in a general equilibrium framework, namely when all markets, for private and public goods, occur simultaneously and interact. The problems that occupy us here are not new, but the solutions proposed here have not been proposed before. The results presented below build on recent results in Chichilnisky (1993a), Chichilnisky and Heal (1994) and Chichilnisky, Heal and Starrett (1993) and Chichilnisky (1996c, 1997a). The main results are the connections that emerge between efficiency and the distribution of property rights in markets with privately produced public goods. This connection represents a major departure from standard markets, in which equilibria are always efficient. Here the distribution of property rights matters. It is decisive in ensuring that the market achieves efficient allocations.

21.2. Efficient ways of privatizing the atmosphere

This section demonstrates that even if the use of the atmosphere is privatized, and the rights to use it are traded in a competitive market, the solutions may not be Pareto efficient, showing how different initial allocations of property rights help achieve efficient allocations. The general equilibrium methodology used in Chichilnisky (1994a) is modified to incorporate these issues following Chichilnisky and Heal (1994) and Chichilnisky, Heal and Starrett (1993).

21.3. Environmental markets

A world market economy in which emission rights are traded along private goods is modeled as follows. There are two regions, North and South denoted by $i = 1, 2$. In each region there is a production function converting energy into private goods,

$$y = \phi_i(a_i),\ i = 1, 2.$$

This leads in each region to a production possibility set exhibiting a tradeoff between more production of consumption goods, y, and a real number a_i measuring abatement or decrease in emissions and thus a better atmosphere and climate:

$$\frac{\partial}{\partial a_i}\phi_i(a_i) < 0,\ \ i = 1, 2.$$

The welfare of each region is represented by a utility function depending on two variables, consumption of private goods y, and the public good a, $u_i: R^2 \to R$, which is an increasing function of both. There is the same level of atmospheric quality in both regions, corresponding to the sum of abatement in both regions denoted, $a = \sum_{i=1}^{2} a_i$:

$$u_i(y_i, a) = u_i\left(y_i, \sum_{i=1}^{2} a_i\right),\ \ i = 1, 2.$$

Each region is assigned initial property rights $\bar{a}_i$, $i = 1, 2$, on the use of the atmosphere so as not to exceed a total internationally agreed ceiling $\bar{a}$:

$$\sum_{i=1i}^{2} \bar{a}_i = \bar{a}.$$

Under this assignation of *property rights*, region i can use an amount of energy (petroleum) that abates $\bar{a}_i$ of carbon dioxide. The amount of energy used by region i is denoted a_i; if the region uses more energy and emits more carbon than its rights allow, i.e. if $a_i \leq \bar{a}_i$, then it has to purchase 'permits' to emit, namely the difference $\bar{a}_i - a_i$; if instead the region uses less energy and emits less carbon than $\bar{a}_i$ it can sell the excess permits to the other region, namely $a_i - \bar{a}_i$.

The price for goods is set equal to one, and the (relative) price of permits is denoted π. The price π is set by market forces as the price that clears all markets, for goods and for permits.

A market equilibrium is defined as follows. Each region $i = 1, 2$ chooses how much energy to use, respectively how much to abate, a_i, with the aim of maximizing utility. The region can afford to consume what it produces plus what it can purchase from the sale of unused permits, or minus what it spends on the purchase of additional permits. Markets are competitive, so the relative price π is taken as a given by each region.[42] Formally: in a market equilibrium each region chooses how much to abate a_i so that to maximize the utility of consumption of y

$$Max(u_i(y_i, a)), \tag{21.1}$$

where the value of consumption y_i satisfies

$$y_i = \phi_i(a_i) + \pi(\bar{a}_i - a_i), \tag{21.2}$$

i.e. y_i equals the value of production plus (minus) the value of exports (imports) of permits. In an equilibrium all markets clear:

$$\sum_i a_i = \bar{a}. \tag{21.3}$$

In summary: the North-South model is defined by giving four parameters and several behavioral rules. The four parameters are property rights and technologies: a_i and ϕ_i, $i = 1, 2$. The equations formalizing the behavior of the economy are 21.1, 21.2 and 21.3. The solutions to these three equations define a market equilibrium for the North-South model, and determine the value of the following variables:

- energy (permits) used by each region, a_i, $i = 1, 2$.
- goods produced by each region, $\phi_i(a_i)$, $i = 1, 2$.
- goods exported (imported) by each region, $y_i - \phi_i(a_i)$, $i = 1, 2$.
- permits imported (exported) by each region, $\bar{a}_i - a_i$.
- price of permits π.
- welfare levels in the two regions, $u_i(y_i, a)$, $i = 1, 2$.

It is now straightforward to verify that although the use of the atmosphere has been privatized, and the rights to use it are traded in a competitive market, the solutions may not be Pareto efficient. The reason is that, due to its physical characteristics, the atmospheric quality a is a public good which is available to all in the same amount, thus linking the consumption of the individuals and presenting a more demanding situation from which to reach efficiency. In particular the Lindahl-Bowen-Samuelson conditions for first best optimality in the allocation of public goods are needed *in addition* to the market equilibrium conditions (21.1),(21.2) and (21.3).[43] In addition to the standard marginally conditions which hold at the market equilibrium[44] the *sum* of the marginal rates of substitution must equal the (common) mar-

[42] Namely the region behaves as if it does not influence prices.
[43] See also Chichilnisky, Heal and Starrett (1993).
[44] The marginal rate of substitution equals the marginal rate of transformation equals relative prices.

ginal rate of transformation which in turn equals the price between a and y, i.e. the permit price π.

21.4. Markets with knowledge

This section presents a general equilibrium model of a market with knowledge.[45] As explained above, knowledge is a privately produced public good. In this sense the model presented below is a model of a market that trades private goods as well as a privately produced public good, in this case, knowledge.

There are two traders, North and South, denoted by the index $i = 1, 2$ respectively, each producing two goods: one private good (x) and another a privately produced public good (a) representing *knowledge*. Each trader h has finite resources (24 hours a day) which are allocated to produce either private goods or knowledge. For each trader $i = 1,2$ there is a tradeoff between producing more private goods and producing more knowledge. However, more knowledge leads to higher productivity. Formally for $i = 1,2$:

$$x_i = g_i(a_i, a), \text{ with } \partial g_i/\partial a_i < 0, \text{ and } \partial g_i/\partial a > 0.$$

where

$$a = \sum_{i=1,2} a_i, \text{ or } a = \sup_{i=1,2}(a_i).$$

Each trader or region has property rights $\Omega_i \in R^2$ on private goods and owns licences that allow them to use knowledge, $a_i \in R$. Traders derive utility from the use of private goods x:

$$u_i(x_i),$$

Through compulsory negotiable licences, knowledge is available to all. Traders may use their licences to access knowledge or may sell their licences in the market. If they wish to use more knowledge than their licences allow, they buy more licences in the market.

Markets for licences are competitive: everyone pays the same price for the same licence; prices are determined by equating supply and demand, and no trader can influence market prices.

The next step is to define a market equilibrium with knowledge. The equilibrium of the market is defined as follows. It consists of

- A price π^*, the relative price between private goods and licences to use knowledge,
- For each trader $i = 1, 2$ a level of initial allocation of property rights on licences to use knowledge in the economy $\bar{a}_1, \bar{a}_2$,

[45] This model is authored by Chichilnisky—it differs from the emissions markets model in several ways, in particular the utility function here does not depend on the public good a as before, and the budget equation below.

- For each trader i a level of consumption of private goods x_i,
- For each trader i knowledge production a_i^*,

so that:

- Each trader i allocates time optimally between the production of knowledge and the production of private goods,
- Each trader maximizes welfare within a budget defined by prices and property rights:

$$Max\, u_i(x_i)$$

$$s.t.\ x_i = g_i(a_i^*, a^*) + \pi^*(\bar{a}_i - a_i^*),$$

i.e. the value of consumption equals the value of production plus the value of licences bought or sold, and

- Markets clear

$$\bar{a}_1 + \bar{a}_2 = a_1^* + a_2^*.$$

A competitive equilibrium determines endogenously a number of prices and quantities:

- the initial allocation of property rights on knowledge in each trader or region;
- the level of production and of consumption of private goods and of knowledge by each trader or region,
- the level of trade of private and knowledge between the parties, as well as
- the terms of trade between the private good and knowledge, π^*, which is the market price of the licences.

The price π^* can be thought of as a market-determined licence fee on using knowledge, since it is a monetary value that must be paid for using knowledge above the level allowed by the initial allocation of property rights.

21.5. Equity and efficiency in environmental markets and in markets for knowledge

The most attractive feature of competitive markets is the efficiency with which they allocate resources, requiring minimal intervention once an appropriate legal infrastructure is in place. This was Adam Smith's vision of the "invisible hand," and was formalized in the neoclassical theory of competitive markets that has prevailed in the Anglo-Saxon world since the 1950's. The efficiency of markets is summarized in *the first welfare theorem of economics*. This theorem establishes that the prices and the allocation of goods and services that arise in a competitive market equilibrium are efficient, in the sense that there is no other allocation that can

make everyone better off. The first welfare theorem has practical importance. It had a major impact in the functioning of economies such as the US, which are market oriented. It underlies much of its *anti-trust legislation*, as well as its *insider trading laws* the laws that restrict *price discrimination*, and other forms of *market discrimination* including gender and age discrimination. The rationale is simple and compelling. Since, according to this theorem, competitive markets ensure an efficient allocation for society, it follows that competitive markets are a "public service." Economic actions that undermine the ability of the market to act competitively therefore detract from the public good.

The insight illustrated here is that the first welfare theorem is no longer valid in markets in which in addition to traditional goods (private goods such as apples or machinery) one trades public goods, such as the rights to use the planet's atmosphere, or knowledge. There is however a new first welfare theorem, reported below as the *first welfare theorem for privately produced public goods*, that establishes that the market reaches efficiency, but only for certain allocations of the rights to use knowledge, or licences. The results are quite general, and apply to any competitive market in which, in addition to private goods, trading involves privately produced public goods. Therefore they apply to environmental markets as well as markets with knowledge. In the case of environmental markets, in the special case considered in those works, the licences involved permits for the use of the atmosphere of the planet as a sink for the emission of greenhouse gases.

Theorem 21.1. *(Chichilnisky, Heal* and *Starrett). Given a total global level of emissions* $\bar{a}$, *there exist* a *finite number of ways to allocate property rights on emissions among the two regions, i.e. there is a finite way of distributing emissions rights (or permits to emit)* $\bar{a}_1, \bar{a}_2$, *with* $\sum_{i=1}^{2} \bar{a}_i = \bar{a}$, *so that at the resulting competitive equilibrium, the allocation of private and public resources in the world economy,* a_1, a_2, x_1, x_2, *is Pareto efficient. For distributions of permits other than these, the competitive market equilibrium is inefficient. When both traders have the same preferences, then the region with more private goods should be given fewer property rights on the public good.*[46]

This theorem is illustrated in diagram 1 provided in the text. The diagram shows a starting distribution of permits that gives proportionately more rights to emit to the North, and computes the corresponding competitive market equilibrium allocation. In a second step, by redistributing the permits in favor of the South and at the same time tightening the emission targets on the whole world, the competitive market achieves a new equilibrium allocation which increases the welfare of the North and the South. This means that the first distribution was not Pareto efficient, and illustrates the potential efficiency gains obtained by redistributing permits in favor of the poorer countries.

[46] For environmental markets rather than markets with knowledge see also Chichilnisky (1993a) Chichilnisky and Heal (1994) and Chichilnisky, Heal and Starrett (1993).

Theorem 21.2. *(Chichilnisky) By allowing total world emissions $\bar{a}$ to vary, one obtains a one-dimensional manifold of property rights from which the competitive market with permits trading achieves a Pareto efficient allocation of the world's resources. For allocations of property rights different from these, the competitive market does not achieve Pareto efficient solutions.*

Proof. See Chichilnisky (1996f and 1997c).

The following result applies to the model presented above, which is different from the model of environmental markets in that the privately produced public good is *knowledge*. The model with knowledge is different from the model of emission markets, because knowledge does not enter in the utility function (as the environmental asset does), but does enter into the production function to improve productivity (as the environmental asset does not).

Theorem 21.3. First welfare theorem of economics for markets with knowledge. *There exists a one-dimensional manifold of property rights allocations from where the market with knowledge achieves an efficient allocation of resources. For allocations of property rights other than these, the competitive market does not achieve Pareto efficient equilibria.*

Proof. See Chichilnisky (1996f, 1997c).

Theorems 21.2 and 21.3 identify the set of all "efficient" allocations of property rights on the use of knowledge, i.e. all allocations of licences to use the available knowledge products in society from which the competitive market achieves efficient allocations of resources as in the case of private goods. It turns out that the allocations that yield efficient solutions provide more property rights to those traders who have fewer property of private goods. As an example, this would involve proving for people to lower income free access to a number of software programs, a number that is larger than the corresponding number would be for someone with larger income.

The rationale behind the proof of these results is simple. Competitive markets in which public goods are traded have more stringent criteria for efficiency than markets for private goods. In addition to the standard marginal conditions (i.e. marginal rates of substitution must equal the marginal rates of transformation) the allocations must also satisfy the Lindahl-Bowen-Samuelson conditions for efficient levels of the public good, requiring that the sum of the marginal rates of substitution equals the (common) marginal rate of transformation between the private and the public good. Since more conditions are needed, the standard competitive allocations are not generally "first best", i.e. they are not generally Pareto efficient. In addition it can be shown that they are not "second best" efficient as well, where second best means that they are Pareto efficient conditional on a total level of world emissions which does not exceed the given target. Generally the total amount of the public good is lower in competitive markets than the "first best" or Pareto efficient level.

21.6. When trade liberalization is helpful to the environment

The following results show how and when trade liberalization fosters efficient use of resources and knowledge, and when it does not. The starting point is Chichilnisky (1993a) where it was shown for the first time how differences in property rights regimes between two regions explain trade, and how the international market amplifies the "tragedy of the commons". In Chichilnisky (1994a) resources are an input of production. This section enlarges the scope of the earlier results showing that there is much more to them than a repetition of the tragedy of the commons; indeed when the ill-defined property rights occur in outputs rather than inputs (e.g. in the case of knowledge goods) the tragedy of the commons is shown to reverse itself. It seems useful to remind the reader of the 1994a methodology in order to facilitate the understanding of the new results presented here.

The methodology used in my 1994a article generalizes 'comparative statics' to more complex and dynamic situations involving property rights.[47] In a nutshell, it consists of comparing the solutions to two sets of simultaneous non-linear equations, each of which represents a North-South economy: one describes a standard competitive market with a unique equilibrium, and the other the same market modified to reflect a historic characteristic of the South, the fact that resources are common property. In some cases they also compare welfare before and after trade.

The 1994a article introduced a mathematical formulation of property rights within the set of equations describing equilibrium and showed how one function, representing resource supplies in the South, varies with the property rights regimes.[48] I showed that at each price more resources are supplied with common property than under private property regimes. This simplified a larger and more diffuse problem by showing that the impact of property rights regimes on the economy can be formalized by the solutions to two sets of non-linear equations describing the North-South economy, differing only in one function: the supply of resources in the South. The first set of equations describes a perfect market equilibrium so that its solutions are Pareto optimal;[49] in addition it has a unique equilibrium. The prices and resource consumptions corresponding to this solution are a benchmark which we use to represent 'optimal prices' and 'optimal consumption' levels in the North and the South. The second set of equations describes an economy where the South has common property rights; it has also a unique equilibrium.

Comparing the prices and the consumption of resources that arise from resolving the first and the second set of equations, the second set has lower resource prices, and higher consumption of resources than the first. From this one deduces that common property of resources leads to lower resource prices, and to more consumption of resources than is optimal. In addition, after trade the welfare of the South is shown to be lower. This is the nature of the results.

[47] A dynamic version of my 1994 results appeared in Chichilnisky (1996d).

[48] It uses a Nash equilibrium solution to a game theoretical model (similar to that in Dasgupta and Heal (1979)) to derive the supply function of resources in terms of prices when the resource pool is common property.

[49] A.k.a. first best optimality.

It is somewhat surprising that the impact of changes in property rights on the solutions to a system of several non-linear equations can be determined simply by examining the slope of one function, the supply function for resources in the South. By revisiting this methodology I show below how it extends to a number of other interesting cases.

The reality is that when recommending resource exports economists have in mind the standard model with private property rights, and my point is that such a model could be misleading. They could be computing apparent gains from trade, not real ones. It is the second model, with common property resources, that is more relevant. In the second model, specializing in resource exports can lead to welfare losses. This is what Chichilnisky (1994a) shows.

21.7. Common property of software

The 1994a methodology can also be used to distinguish the results from the so-called 'tragedy of the commons,' namely the tendency of societies to overuse their common property resources. There is an interesting analogy here, but I show below that with common property regimes international trade can lead either to overconsumption or to underconsumption of resources. The former case confirms the tragedy of the commons, but the latter goes in the opposite direction. Matters are more complex and subtle than a mere extension of the tragedy of the commons to the international arena. The following result explains what makes the difference.

As pointed out above, a crucial factor emerging from my 1994a results is the form of the supply function for resources under two different property rights regimes. Recall the simplest case analyzed in my Chichilnisky (1994a) article. A number of harvesters in the South extract a resource (such as fish or trees) from a pool that is owned as *common property* (such as a lake or a forest), using their labor, x. I showed that at any market price p the harvesters apply an amount of effort x which optimizes the difference between average revenues and marginal cost (Chichilnisky (1994a), see also Dasgupta and Heal (1979)), so that x satisfies the expression.

$$pf(x)/x = c'(x).$$

Matters are different under private property regimes; here, instead, the harvesters optimize profits:

$$Max_x[pf(x) - c(x)],$$

and the optimal amount of effort x therefore satisfies the expression

$$pf'(x) = c'(x).$$

Since the production function $f(x)$ is concave, it follows that for all x, average productivity exceeds marginal productivity $f(x)/x > f'(x)$, so that the rewards appear to be larger than they are and the optimal amount of effort x applied in the former case is larger than in the second. Therefore more resources are extracted with

common property than with private property at any price p, and the supply function for resources has a higher slope[50] with common property than it does with private property. In a nutshell: the problem of overextraction of resources arises because the pool from which the resource is extracted, namely the *input* used to 'produce' fish or wood, is common property,

Matters are quite different when an *output* of production is common property. Software can provide a good example. An economy produces software using labor x and (privately owned) capital. Software is an output that is sold to a number of commercial clients who pay a market price p; it is however treated as common property by non-commercial users. A proportion k, $0 < k < 1$, of the users are commercial, the others are not. Once the software is produced there is no cost for duplicating the software, and no restrictions or penalties for non-commercial users in doing so. Using the 1994 method of analysis, we focus on the shape of the supply function for software in one region under two regimes: private property and common property.

Under private property rights, all users pay for the software, so the workers maximize the profits function:

$$Max_x\,[\,pf(x) - c(x)],$$

choosing therefore their input x so it satisfies the standard conditions:

$$pf'(x) = c'(x).$$

However, with common property regimes for non-commercial users, the worker's benefits are instead

$$\frac{1}{k}pf(x) - c(x),$$

so that workers choose effort levels x that maximize benefits, i.e. those satisfying

$$\frac{1}{k}pf'(x) = c'(x).$$

Since $pf'(x) > \frac{1}{k}pf'(x)$, and f' is a decreasing function because f is concave, when software is common property among non-commercial users, less effort x is applied to production than under private property regimes. The supply function for software as a function of p has a *lower* slope with common property than with private property. In a general equilibrium *less* software is produced at any price. By contrast, in the case of resources the common property supply function has a *higher* slope in the case of common property rights, and *more* resources are produced at each price.

Furthermore, if there were no penalties associated with duplicating software by any user ($k \sim \infty$), as it is in countries with weak intellectual property legislation such

[50] As a function of prices.

as Japan or China, there would be no effort applied to producing software except for that which the producer itself can benefit from. The supply function would be nearly zero. Software is a labor-intensive commodity that uses skilled labor, which is a comparative advantage in a nation such as China or Japan. These countries would therefore be expected to export software under standard assumptions. Nevertheless the results given above would predict that these countries would not specialize in software, importing more software than is optimal, namely more software than what they would import under a private property rights regime. The facts agree with this observation: neither Japan nor China have developed a software industry despite their comparative advantages in abundant skilled labor. Applying the rest of the methodology developed in my 1994 paper to this newly derived supply function for software, one can deduce that China and Japan would be better off if they were to develop a better regime to protect intellectual property rights, running counter to their current policies.

For the purposes of this paper the main observation emerging from this analysis is that common property can lead to *overproduction* or to *underproduction*. Using the same methodology I introduced in 1994 one can deal with a variety of different examples. The results presented then are more than an extension of the tragedy of the commons.

The following subsection explains when taxes or tariffs may not be adequate policies.

21.8. Tariffs

Tariffs or taxes could depress the price of resources or agricultural output, and therefore may induce a reallocation of labor away from extracting or deforesting activities. This is the rationale for introducing barriers to trade. However, as I showed in 1994, tariffs that depress the price of resources may have the opposite effect to that desired when there is a binding subsistence constraint, and therefore they are not reliable policy. Margolis (1997) discusses trade barriers and comes to a similar conclusion. Cruz and Repetto (1992) have reported interesting empirical work supporting this conclusion as well, on the effects of structural adjustment polices in the Philippines. They discuss the 'income effects' of adjustment, which have precisely the effect I predicted theoretically on exploitation of forests and coastal fisheries. Repetto[51] points out, however, that where there are still substantial resource rents, as in commercial logging of tropical timbers, the rent capture through resource taxes can have the opposite effect of reducing the rate of extraction.

[51] Private communication, October 1994.

References

Atkinson, A. and Stiglitz, J. E. (1980) *Lectures on Public Economics* (Maidenhead, England: McGraw Hill).

Barrett, S. (1991) "Economic Instruments for Climate Change Policy" Chapter 2 in OECD: "Responding to Climate Change: Selected Economic Issues" Paris, 1991, p. 51-109.

Barrett, S. (1992) "Acceptable Allocations of Tradeable Carbon Emission Entitlements in a Global Warming Treaty" Chapter VI in UNCTAD (ed.) Combatting Global Warming; Study on a Global System of Tradeable Carbon Emission Entitlements, New York, United Nations, 1992, p. 85-125.

Beltratti, A., G. Chichilnisky and G. M. Heal (1992) "Option Markets and Environmental Irreversibilities", First Boston Working Paper, Columbia Business School, New York.

Beltratti, A., G. Chichilnisky and G. M. Heal (1995) "The Green Golden Rule", *Economic Letters*, Vol. 49, No. 2, pp. 175-180, August 1995.

Bernardez, C. (1996) "Learning Curve: Environmental Assets and Derivatives", *Derivatives Week, Institutional Investor*, 3 June, 1996, Vol. V, No. 22.

Bruce, J., H. Lee and E. Haites (1995) "Climate Change 1995: Economic and Social Dimensions of Climate Change: Contribution of Working Group III to the Second Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Chapter 11, "An Economic Assessment of Policy Instruments for Combatting Climate Change" Lead Authors: B. Fischer, S. Barrett, P. Bofm, M. Kuroda, J. Mubazi, A. Shah, R. Stavins.

Brundtland, G.H. (1987) *The UN World Commission on Environment and Development*, Oxford: Oxford University Press.

Chichilnisky, G. (1977a) "Economic Development and Efficiency Criteria in the Satisfaction of Basic Needs", *Applied Mathematical Modelling*, VoL 1, No: 6, pp. 290-297.

Chichilnisky, G. (1977b) "Development Patterns and the International Order" Special Issue on the International Economic Order, *Journal of International Affairs*, 1977.

Chichilnisky, G. (1981) "Terms of Trade and Domestic Distribution: Export Led Growth with Abundant Labor", *Journal of Development Economics* 8, pp. 163-192.

Chichilnisky, G. (1993a) "The abatement of CO_2 emission in industrial and developing countries", OECD/IEA conference on "The Economics of Climate Change", published in *OECD: The Economics of Climate Change*, (ed. T. Jones), Paris, June 1993, pp. 159-170.

Chichilnisky, G. (1993b) "North-South Trade and the Dynamics of Renewable Resources", *Structural Change and Economic Dynamics*, Oxford University Press, December 1993, Vol. 4, No. 2, pp. 219-248.

Chichilnisky, G. (1993c) "Property Rights on Biodiversity and the Pharmaceutical Industry", Case Study, Columbia Business School.

Chichilnisky, G. (1994a) "North-South Trade and the Global Environment." *American Economic Review*, Vol. 84, No. 4, September 1994, pp. 427-434.

Chichilnisky, G. (1994b) "Trade Regimes and GATT: Resource Intensive vs. Knowledge Intensive Growth" *Economic Systems* merged with *Journal of International Comparative Economics*, Special Issue on Globalization of the World Economy, CIDEI Conference, Rome, 1994, 20, 1996, pp. 147-181.

Chichilnisky, G. (1995a) "Strategies for Trade Liberalization in the Americas", in *Trade Liberalization in the Americas*, Interamerican Development Bank (IDB) and United

Nations Commission for Latin America and the Caribbean (ECLAC), Washington, D.C.

Chichilnisky, G. (1995b) "Global Environmental Markets: The Case for an International Bank for Environmental Settlements", *Proceedings of the Third Annual World Bank Conference on Effective Financing for Environmentally Sustainable Development*, World Bank, Washington, D.C. 6 October, 1995, UNESCO-UNDP Technical Report No. 10, Office of Development Studies (ODS) UNDP, New York, N.Y, 10017, September 1996.

Chichilnisky, G. (1995-1996) "The Economic Value of the Earth's Resources", Invited perspective article, *Trends in Ecology and Evolution*, (TREE) 1995-1996, pp. 135-40.

Chichilnisky, G. (1996a) "An Axiomatic Approach to Sustainable Development", *Social Choice and Welfare*, Vol. 13, No. 2, pp. 357-364, first circulated as "What is Sustainable Development?" Technical Report No. 65, Stanford Institute for Theoretical Economics, 1993.

Chichilnisky, G. (1996b) "The Greening of Bretton Woods", *Financial Times*, Section on Economics and the Environment, 10 January, 1996, p. 8.

Chichilnisky, G. (1996c) "Efficient Property Rights in Markets for Knowledge", Invited Address to the American Economic Association in the AAERE meetings, San Francisco, January 1996).

Chichilnisky, G. (1996d) "Property Rights and the Dynamics of North-South Trade", Chapter 8 in *Agriculture, Trade and Environment* (eds. M. Bredahl, N. Ballenger, J. Dunmore and T. Roe) Westview Press, Harper Collins Publishers and Oxford, 1996, pp. 97-110.

Chichilnisky. G. (1997a) "The Knowledge Revolution: Its impact on Consumption Patterns and Resource Use", Human Development Report, United Nations Development Program (UNDP) New York, November 1997.

Chichilnisky, G. (1997b) "Updating Property Rights for the Knowledge Revolution", John D. and Catherine T. McArthur Lecture, Program on Multilateralism, Institute for International Studies, University of California, Berkeley, November 3, 1997.

Chichilnisky, G. (1997c) "Markets with Knowledge" Keynote Address, Industrial Economics Workshop, Business School, University of Melbourne, Australia, July 1997, to appear in the Proceedings of the Industrial Economics Workshop, 1998.

Chichilnisky, G. and G. Heal. (1993a) "Global Environmental Risks", *Journal of Economic Perspectives*, Special Issue on the Environment, Fall, 1993, pp. 65-86.

Chichilnisky, G. and G. Heal. (1993b) "Sustainable Growth and the Green Golden Rule", in *OECD: Economic Approaches to Sustainable Development* (eds. I. Goldin and A. Winters), Paris, May, 1993, pp. 147-165.

Chichilnisky, G. and G. Heal. (1994) "Who should abate carbon emission? An international perspective", *Economics Letters*, Spring 1994, pp. 443-449.

Chichilnisky, G. and G. Heal. (1995) "Tradeable Carbon Emissions: Principles and Practice", Report to Working Party No.1 OECD, OECD Economics Working Paper, Paris.

Chichilnisky G. and G. Heal. (2000) *Environmental Markets: Equity and Efficiency*. Columbia University Press, Series on Economics for a Sustainable Earth.

Chichilnisky, G., G. Heal and D. Starrett. (1993) "International Markets with Emission Rights: Equity and Efficiency", *Center for Economic Policy Research*, Publication No. 81, Stanford University, Fall 1993.

Coppel, J. (1993) "Implementing a global abatement policy: some selected issues", OECD/IEA conference on "The Economics of Climate Change", Paris, June 1993, published in *OECD: The Economics of Climate Change* (ed. T. Jones), Paris, June 1993.

Copeland, B. and S. Taylor (1994) "North South Trade and the Environment" *Quarterly Journal of Economics* 109: 775-787.

Cruz, W. and R. Repetto (1992) The Environmental Effects of Stabilization and Structural Adjustment Programs: the Philippines Case" *World Resources Institute*, Washington, D.C.

Dasgupta, P. and G. Heal (1979) *Economic Theory and Exhaustible Resources*, Cambridge University Press, Cambridge.

Fidler, S. (1995) "An Information Age Society is Booming," *Financial Times*, 26 April, 1995.

Harris, D. J. (1994) "Determinants of Aggregate Export Performance of Caribbean Countries: A Comparative Analysis of Barbados, Costa Rica, Dominican Republic, Jamaica and Trinidad & Tobago", Department of Economics, Stanford University, September 1994.

Heal, G. (1991) "Economics of climate change." In R.E. Just and N. Bockstahl, *Commodity and Resource Policies in Agricultural Systems*, Springer Verlag, Heidelberg.

Heal, G. (1993) "International Negotiations on Emission Control." *Structural Change and Economic Dynamics*.

Heal, G. (1999) *Valuing the Future*, Columbia University Press, Series on Economics for a Sustainable Earth (eds. G. Chichilnisky and G. Heal).

Heal, G. and G. Chichilnisky (1991) *Oil and the International Economy*. Clarendon Press, Oxford University Press.

Laffont, J-J. (1977) Effets Externes et Theorie Economique, Monographes du Seminaire d'Econometrie, publie sous la direction de Rene Roy et Edmond Malinvaud, Editions du Centre National de la Research Scientifique, 15, quai Anatole France, 75700 Paris, 1977.

Low, P. (ed.) (1998) "International Trade and the Environment," *World Bank Discussion Paper*. No. 159, The World Bank, Washington D.C. 1998

Margolis, M. (1997) "North-South Trade and the Global Environment: A Comment" Working Paper, Department of Economics and Finance, University of Wyoming, April 1997.

Tietenberg, T. (1985) *Emissions Trading*, Resources for the Future, Washington, D.C.

World Bank (1992) World Development Report, Oxford University Press 1992.

World Resources: People and the Environment (1994-1995) A Guide to the Global Environment. Report by the World Resources Institute, the United Nations Environment Program and the United Nations Development Program, Oxford University Press.

World Resources Institute (1994-5) *World Resources 1994-5, a Guide to the Global Environment*, Oxford Press.

World Wide Fund for Nature (1998) The UN Biodiversity Convention and the WTO TRIPS Agreement: Recommendations to Avoid Conflict and Promote Sustainable Development, WWF International Discussion Paper.

Yamin, F. "Biodiversity Conservation and Intellectual Property Rights", Position Paper, World Wide Fund for Nature, September 12, 1994.

Synthesis and Future Directions

Capturing the Value of Ecosystem Services to Protect Biodiversity

Walter V. Reid
World Resources Institute,
10 G St. NE,
Washington, D.C. 20002 USA
Current address: 731 N 79th St.,
Seattle, WA 98103
waltreid@attglobal.net

Abstract

Ecosystem services have considerable economic value. This paper explores the potential that the economic value of ecosystem services has for financing conservation. More specifically, it examines the potential for capturing the economic value of such ecosystem services as water quantity and quality, carbon sequestration, pollination and biocontrol, waste treatment, and flood and storm protection. Where institutional mechanisms do appear to exist that would enable some portion of the economic value of the service to be captured, I then examine the scope for financing conservation of the service through the economic value that can be captured and make very rough estimates of the amount of land that might be conserved to maintain the service. For ecosystem services associated with water quantity and quality, significant scope for action exists. Management of more than 10% of land area for its watershed value for municipal water supplies could currently be justified globally and this could double by the middle of the next century. Management for carbon sequestration could be a goal for as much as 5–13% of tropical forest area. For a number of other ecosystem services, including pollination, pest control, waste treatment, and flood and storm protection, opportunities do exist to capture some portion of the value associated with these services, but the scope for using such mechanisms as a source of financial support for conservation of the service appears to be far more limited than is the case with water and carbon services.

Introduction

Can conservation pay? Can we conserve biodiversity if it doesn't pay? Few questions can spark a more heated debate among conservationists than these. For some, economic justifications and market incentives for conservation are tantamount to a death knell for biodiversity. For others, they are the only hope in the face of inexorable pressures to convert natural habitats to more economically profitable uses.

A strong case can be made that a conservation strategy relying solely on economic values will be self-defeating (Sagoff, 1996). The market value of a region's biodiversity is typically much lower than alternative uses of the region's forests, land, or water, even when "externalities" can be incorporated (which they rarely can).[1] And even though markets sometimes work in biodiversity's favor, potentially more powerful moral or ethical justifications for conservation can be undermined by too great a focus on economic opportunities. By way of analogy, our society concluded that child labor was wrong as a matter of principle, not because of cost-benefit analyses (which probably would support the economic value of child labor) (Sagoff, 1996). In addition, the track record of market-based incentives for conservation is mixed. It has often been the very components of biodiversity with the highest market value, such as timber, fish, and medicinal plants that have been depleted most rapidly. Finally, the presumption that conservation must pay for itself can be in conflict with the common "polluter pays principle" used to allocate responsibility for environmental impacts (Chomitz et al., 1998).

But if it is risky to link the future of biodiversity entirely to cost-benefit calculations, it would be an even greater mistake to wage the battle for biodiversity conservation on moral grounds alone. There is tremendous scope for correcting economically wasteful market and policy failures related to natural resource management. These changes are justified on economic grounds but can have, as an "unintended" effect, significant benefits for biodiversity as well. For example, it makes good economic sense to end below-cost timber sales and stop subsidizing cattle grazing on public lands, and these actions would also lessen pressure on biodiversity. Furthermore, some industries depend on the protection of biodiversity. For example, the maintenance of a coastal fishery may require protection of the spawning grounds for the fish, and this protected area will harbor many other species as well. Similarly, ecotourism—a major industry in countries like Costa Rica, Kenya, and Ecuador—requires the protection of wildlands.

These opportunities for actions justified by the market that have "side benefits" of biodiversity conservation are the "low-hanging fruit" of biodiversity conservation, the elusive "win-win" solutions. These actions align private benefits with public benefits and thus help to achieve a desirable public goal through market economic forces. They will not be enough to stop the global loss of biodiversity,

[1] In this paper, "biodiversity" refers to the entire range of genetic, species, and ecosystem diversity in a region. Only a small fraction of a region's biodiversity, referred to as "biological resources," are traded in markets. Biological resources would include fish and wildlife that are harvested for food or trees cut for fuelwood or timber.

but they can slow the loss of diversity using the strength and momentum of market forces rather than having to overcome those forces.

Over the past decade, the conservation community has explored the potential for a number of economically profitable wildlands goods and services to help lessen pressure on biodiversity. Conservation projects are now underway that seek to create economic incentives for biodiversity conservation through ecotourism, nontimber forest products, game ranching, and bioprospecting. In addition, new resource management techniques, such as natural forest management, have been developed in an effort to increase the economic profitability of forms of resource exploitation that are less destructive of biodiversity. These efforts promise to reduce pressure on biodiversity. But there is potentially an even larger source of economic value associated with natural ecosystems that has not yet been tapped as a means of creating economic incentives for biodiversity conservation: the value

Box 1. Ecosystem Services and Functions.

Service	*Function*
Gas regulation	Regulate atmospheric chemical composition
Climate regulation	Regulate global temperature, precipitation, and other biological mediated climatic processes at global and local levels
Disturbance regulation	Provide storm protection, flood control, drought recovery, and other responses to environmental variability.
Water regulation	Regulate hydrological flows
Water supply	Store and retain water
Soil formation	Support soil-formation processes
Erosion control	Retain soil within an ecosystem
Nutrient cycling	Store, internally cycle, process, and acquire nutrients
Waste treatment	Enable recovery of mobile nutrients and removal or breakdown of excess nutrients and compounds
Pollination	Support pollinators for reproduction of plant populations
Biological control	Enable trophic-dynamic regulation of populations
Refugia	Provide habitat for resident and transient populations
Food production	Convert a portion of gross primary production into food
Raw materials	Convert a portion of gross primary production into raw materials
Genetic resources	Produce unique biological materials and products
Recreation	Provide opportunities for recreational activities
Cultural resources	Provide opportunities for non-commercial activities

Source: Costanza et al. (1997).

of ecosystem services. Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life (Daily, 1997) (see Box 1). A variety of ecosystem services, such as the protection of water quality and quantity, pollinator services, flood control, and pest control have substantial economic value, which could, in theory, justify public or private actions to protect those services. And in many instances, protection of those ecosystem services would also lessen pressure on biodiversity.

This paper examines the potential that the economic value of ecosystem services holds for financing conservation. In other words, to what extent can we capture the value of ecosystem services in order to finance the protection of that service? A recent (and highly controversial) study estimates the value of ecosystem services worldwide to be $33 trillion per year, 20% greater than global Gross National Product in 1995 (Costanza et al., 1997; but see Sagoff, 1997; Pearce, 1998). Another analysis by Pimentel et al. (1997) reached a lower, but still substantial, estimate. Although these services have immense value to society, they are not usually traded in markets and typically have no market value. Nobody captures the full economic benefit of protecting the service, but many profitable actions such as forest harvest, wetland conversion, or the discharge of untreated wastes into rivers can degrade or destroy these services. Consequently, like other public goods, ecosystem services tend to diminish beyond an optimal level for our needs despite their economic value.

The substantial economic value of ecosystem services is informative, but has little practical consequence unless some mechanism exists to capture that value and to create an economic incentive to maintain the service. By way of analogy, the fact that someone in the United States would express a "willingness to pay" to prevent the extinction of lions in Kenya won't automatically cause policymakers in Kenya to establish a lion reserve. But if that interest in lions results in greater tourism, and Kenya can profit from that tourism, then a conservation incentive will exist.

What is the scope for policies or institutions that can create incentives for stewardship of those services (and their associated biodiversity) by capturing value of the service? Of the services listed in Table 1, this paper examines a subset that have received relatively little attention but which seem to hold the potential for people to pay to protect the service. Thus, I exclude food production, raw materials, genetic resources, and recreation because of the attention they have received elsewhere (e.g., Myers, 1984; McNeely et al., 1990; Reid et al., 1993). I also exclude cultural services, defined by Costanza et al. (1997) as "providing opportunities for non-commercial uses," which by definition would not provide commercial incentives for conservation. Of the remaining services, I focus on those where people today would be willing to pay to maintain the service. For example, if the service is growing scarce and its loss would create short- or long-term economic costs to some users, people might pay to protect it. Thus, since the loss of natural pest control services would require farmers to pay more for pesticides, or loss of the carbon-sequestration service of forests could result in greater economic damages from climate change, people might be willing to pay to protect the service. In

contrast, oxygen production and soil formation are critical services provided by ecosystems but neither is growing scarce. (Soil is growing scarce, through erosion, but typically the process of soil generation itself is not being diminished.) Thus, despite their economic value, people would have little motivation to pay for their protection.

The resulting set of ecosystem services explored in this paper includes: (I) water quantity and quality, (II) carbon sequestration, (III) pollination and pest control, (IV) waste treatment, and (V) flood and storm protection.

I. Water Quantity and Quality

People have long appreciated the importance of protecting watersheds in order to protect water quality and quantity. More than 150 years ago, for example, New York City took steps to slow development of the Catskill Mountains and Croton

Table 1. Urban water supply watershed areas.

Urban region	Population	Watershed	Hectares	Ha per capita	Source
Massachusetts (46 towns)	2,313,000	394 sq mile Quabbin, Ware, and Wachusett	102,046	0.044	Mass. Water Res. Authority
New York City*	9 million	2000 sq mile Catskills, Croton	518,000	0.58	
New Jersey	2 million	17,300 acres Sterling Forest	7,006	0.004	
Seattle, Washington*	1.2 million	103,885 acres Cedar River and S. Fork Tolt Watershed	42,073	0.035	Seattle Public Utilities
Portland, Maine*	160,000	436 sq miles Sebago Lake watershed	112,924	0.70	Portland Water Dist.
Portland, Oregon*	825,000	106 sq mile Bull Run Watershed	27,454	0.033	Portland Water Bureau
Providence, RI*	600,000	60,000 acres	24,300	0.040	Providence Water
Cape Town, South Africa	2.2 million	617,750 acres	250,188	0.114	
Santa Barbara, CA	100,000	294,400 acres	119,232	1.19	
Bellingham, WA	65,000	73,984 acres	29,963	0.46	

* = unfiltered water supply systems; note: New York filters 10% of its supply

Watershed in order to protect the City's water supply. During the early 20th century, the City of Los Angeles purchased water rights and land in California's Owens Valley—some 500 miles away—to gain and protect access to that water supply. What policy and institutional mechanisms can be used to capture values associated with water and to finance watershed protection? And, given these mechanisms, what is the scope for further watershed protection to protect these values?

Mechanisms for capturing ecosystem service values associated with water

New York City's recent decision to invest in watershed protection instead of constructing a water filtration plant is one of the best-known examples of ecosystem service protection linked to the value of that service (Chichilnisky & Heal, 1997). The Catskill Watershed and the Croton Watershed—located about 100 miles and 50 miles from New York City, respectively, and totaling 1900 square miles—provide drinking water for the city's 8 million customers and 1 million others in surrounding communities. New York City has used these sources without filtration for more than 150 years. In the early 1990s, the City faced a challenge when new EPA regulations implementing the Safe Drinking Water Act required that surface drinking water supplies must be filtered if they could not meet new water quality standards by June 1993. Development in the Croton Watershed, which provides 10% of the water supply, had already degraded the watershed to the point that filtration was required. But for the Catskills, the City faced a choice between filtration and watershed protection.

The estimated cost for filtering the Catskill supplies was some $4 to $6 billion, plus $300 million in annual operating expenses. This would require a doubling of the City's water rates (Budrock, 1997). Instead, New York sought to protect its water quality through watershed management. The City obtained a "Filtration Avoidance Determination" from EPA, and, after several years of negotiation, reached agreement with communities in the Catskills, New York State, and the EPA on a Watershed Protection Plan. Under this plan, the city, state, and federal governments commit $1.4 billion to improve water supplies through such means as watershed protection, infrastructure and water quality improvements, and partnership programs with local communities (State of New York, 1998). Of this total, the City itself committed $1.216 billion, including $666 million for work in the watershed, of which $250 to $300 million will be used for acquisition of up to 355,000 acres of eligible land (to be acquired on a voluntary basis). This agreement was financed through a $1.75 billion bond initiative passed in November 1996. Under this plan, a typical New York City water bill will increase only 9%, from $400 per year to $435 per year in 2002 (Revkin, 1997).

The New York City case is notable for the magnitude of investment in land conservation to protect this ecosystem service and the magnitude of the avoided cost, but it is just one of many examples of programs or policies designed to finance the protection of watersheds by capturing a portion of the value of the wa-

ter provided. The various mechanisms can be divided into two general categories: (1) those relying on broad-based sources of financing (typically general tax revenues), and (2) those that more directly link watershed conservation to a fee or tax associated with the use of water.

1. Financing watershed protection with broad-based public revenues

Many national parks and other kinds of protected areas around the world have been established (or justified) in part because of the protection they provide for water quality or quantity (see Box 2). Typically, the funds for land purchase and management of these protected areas have come from general revenues, not from the value of the water that they provide. Technically, the use of general funds to protect an ecosystem service such as water quantity or quality is a second-best arrangement, since all users pay the same amount to protect the service even though some gain greater benefits than others. For example, a farmer might use vastly more water than an urban resident, yet each user would pay the same for the service. In contrast, if funds for watershed protection came from water use fees, individuals using more water would have an incentive to conserve water and would pay a greater share of the costs of protecting the service. But though this approach of using broad-based financing is economically distorting, it is widely used around the world, and does make sense for protected areas that provide multiple social benefits beyond watershed protection. To the extent that a protected area is dependent on user fees, incentives will exist for managers to favor the attributes generating those fees, such as tourism or water production, with inadequate investment in resource management.

Within the United States, for example, a substantial amount of watershed protection is occurring as part of larger "open space protection" initiatives typically funded by income or property taxes. When it comes to green space protection, the

Box 2. Examples of watershed protection justification for protected area establishment.

- The 7600-ha La Tigra National Park in Honduras provides the capital city of Tegucigalpa with 40% of its drinking water at a cost of about 5% of the second-largest source.
- Guatopo National Park in Venezuela provides 20,000 liters per second of high-quality water to Caracas; $25 million has been spent buying out timber and farming interests in the area.
- 13 of Venezuela's 39 national parks protect urban water supplies, contributing some 60% of the urban population. The replacement cost of the water lost if these areas were deforested is estimated to have a net present value of $103 million to $206 million.
- Dumoga-Bone National Park in Indonesia was established with a World Bank loan of $1.2 million, justified on the basis of the protection the park provides to a major irrigation project in the lowlands.
- Maduru Oya National Park in Sri Lanka protects the watersheds of four reservoirs, including the major reservoir for the Mahaweli Development Project.

Source: McNeely (1998); Gutman (1995); Garcia (1984, cited in McNeely (1998)).

American public is much more willing to pay new taxes than conventional wisdom suggests. In 1996, voters approved 70% of more than 150 conservation-finance ballot initiatives, amounting to over $4 billion for state and local funding for parks, open space, and environmental revitalization, and the approval rate grew to 80% in the following year (Myers, 1997a, 1997b). With many state budgets operating in the black for the first time in decades, open space protection is proving politically popular as well. The Governor of Connecticut, for example, recently proposed a plan to preserve 21% of Connecticut as open space by 2023—an increase of more than 200,000 acres—relying on $160 million of state funding over the first 5 years.

The Ecological Value Added Tax (VAT) in the State of Paraná, Brazil, is one innovative example of the use of general public revenues to finance watershed protection (Vogel, 1997). Paraná relies on the VAT for more than 90% of tax revenues. During the 1980s, the Instituto Ambiental do Paraná (IAP), a state agency, succeeded in changing the criteria for distribution of these tax revenues, earmarking 5% for the environment, of which half goes to municipalities that maintain public watersheds. Municipalities receive funds in proportion to the relative progress they are making to improve water quality in the conservation units that they manage. Between 1991 and 1995, water quality improved by 68% in sites monitored by IAP. Some 71 rural communities benefit from this program, and it has become a major revenue source for some. Proceeds from the VAT now account for 85% of the municipal budget of Piraquara, which serves as the watershed for the city of Curitiba.

The "Working for Water" program in South Africa is another example of broad-based financing of watershed protection. Water, a scarce resource in South Africa to begin with, is becoming even more limited as a result of the spread of water-

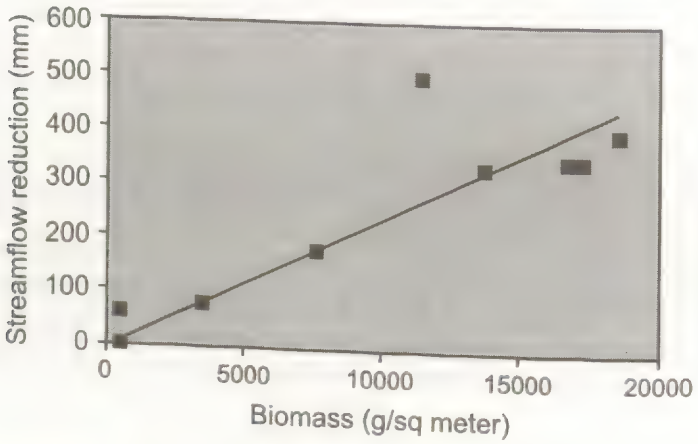


Figure 1. Relationship of aboveground biomass to streamflow reduction in the fynbos region of South Africa (Le Maitre et al., 1996).

demanding species of alien plants such as eucalyptus, pine, and black wattle (*Acacia mearnsii*) introduced from other countries. The native fynbos (shrubland) ecosystems in the Western Cape province are adapted to summer droughts, and the relatively low biomass in these ecosystems ensures conservative water use. But the fynbos is highly susceptible to invasion by introduced woody species, and the 3- to 10-fold increase in biomass associated with these species places severe demands on water resources, significantly reducing runoff (Le Maitre et al., 1996) (Fig. 1). One study calculated that the invasion of Cape Town's watershed by alien species would result in an average decrease in water discharge of 347 m^3/ha/yr over the next 100 years, equal to a loss of 30% of the city's water supply on average and a loss of more than 50% of supply in some years. The cost of removing the introduced species by hand amounts to between $140 and $830 per hectare, with annual maintenance costs of an additional $8/ha. The increased runoff resulting from the removal of alien species, however, more than offsets these costs. Watersheds where alien species are removed can produce water at a unit cost of 11.9cents/m^3 (including costs of removal of alien species), while the unit cost in watersheds with alien species is 13.8cents/m^3 (Le Maitre et al., 1996).

The Working for Water program was established to protect water supplies in the Cape region by removing introduced species by hand, and this program is now in place across the country. The program targets some 160 different species for eradication, including eucalyptus and black wattle from Australia, various European and American pines including Monterey Pine from California, *Sesbania*, and *Prosopsis* (also a major problem in the Southwestern United States). The program's goals are to: (i) increase water yield, (ii) conserve biological diversity, (iii) create employment, and (iv) enhance public awareness about water conservation. The biodiversity in the fynbos is extremely rich; the Cape flora has ca. 8574 species of vascular plants, with 68% endemic to the region (van Wilgen et al., 1996). Because alien species are a significant threat to native biodiversity, the eradication program has direct benefits for biodiversity conservation. In 1996–1997, the Working for Water Program spent R77 million (US$15 million) and employed some 8300 low-skilled workers (Dept. Of Water Affairs and Forestry, 1997). The program is expected to grow to a budget of some $50 million with 35,000 people employed in 1998 (G. Preston, pers. comm. 1998).

2. Linking financing of watershed protection to water use

By more directly tying the financing of watershed protection to the economic benefits provided by the water, economic efficiency can be increased. Moreover, despite the public's willingness to pay higher taxes to support open-space protection, there are many competing uses for general public revenues, such as education and health care, and thus it is well worth searching for dedicated revenue streams that could complement existing programs.

Many water and power utilities around the world protect the watersheds they rely upon as part of their general operations, although the extent of watershed conservation varies considerably. In the United States, for example, water utilities

own approximately 2% of the watersheds providing urban drinking water supplies. These land conservation programs have traditionally been funded out of the general water or electricity charge, rather than with a separate tax or surcharge. For example:

- In Venezuela, the Caroni river basin is the catchment for 95% of the country's present and planned hydro-development, which supplies 72% of the national power production and is the second largest hydropower system in the world. Recognizing that the vegetative cover of the watershed needed to be maintained to avoid siltation of the reservoirs, the utility operating this hydro system lobbied for extension of protected areas (particularly Canaima National Park) and a legal ban on commercial agriculture in the basin. The utility also operates and pays for a watershed surveillance program, fire control, and contributes to health care and environmental education programs for communities in the basin—activities amounting to $10.7 million each year (equal to half of Venezuela's entire Park Service budget) (Gutman, 1995).
- Bonneville Power Administration, which operates hydropower plants on the Columbia River in the United States, allocates several million dollars per year to wildlife conservation activities, particularly related to its efforts to restore populations of anadromous fish such as salmon.
- The city of Portland, Maine, spends $729,000 annually (administration, monitoring, programs, transportation) on watershed protection. As a result of its program, it has received an EPA filtration waiver, resulting in avoided costs of $25 million for construction of a filtration facility and an additional $750,000 annual operating costs.
- Portland, Oregon, spends $920,000 per year for watershed protection (96% of the watershed is owned by the U.S. Forest Service). By maintaining the unfiltered status of the water, the City has avoided an estimated $200 million capital cost for a filtration plant (Portland Water Bureau, 1997).
- Watersheds providing drinking water cover more than 16% of the state of Connecticut. Since the mid-19th century, Connecticut water companies have acquired more than 130,000 acres of land around drinking water supplies, equal to more than half of the state's open space holdings. The state's source protection efforts have resulted in waivers worth more than $19 million in reduced water monitoring costs over recent years (Trust for Public Land, 1998).
- Energía Global, a private hyrdo-electricity provider in Costa Rica, offers landowners in its watersheds $10/hectare/year to maintain or restore forest cover on their plots (Chomitz et al., 1998).

In other cases, cities or utilities have imposed specific taxes or surcharges to fund watershed protection and land conservation. For example, Dade County, Florida, imposes a 3% surcharge on water bills for watershed protection, and Spokane, Washington, residents pay $15 per year for aquifer protection (Trust for Public Land, 1997). In 1989, Rhode Island passed legislation establishing a 2.59 cents

per 100 gallon surcharge on water use. Roughly half of the revenues support various state water programs, but the remaining 1.25 cents per 100 gallons are retained by the utilities, which are required to spend at least 55% on land acquisition. Under this program, the City of Providence has expanded the ownership of its watershed by 1500 acres since 1989 to a total of 17,000 acres, or 28% of the 60,000-acre watershed. Overall, Providence has spent $16 million on land purchases over this period (the legislation allows the utilities to borrow up to $10 million from the state, which is then paid back as fees are collected). Providence is undertaking these actions even though it already has in place a standard filtration and treatment facility.

In each of these cases, decision-makers have justified the cost of watershed protection in part because it is lower than the cost of the loss of that service. In other words, if the goal is clean water for urban consumers or reduced siltation to increase the life-span of a hydroproject, watershed protection is often the investment that will give the greatest water quality benefits per dollar invested.

Another innovative approach to watershed protection is being tested in the Pacific Northwest of the United States. Watershed protection helps to ensure that water is available for downstream uses. But because some of those uses require withdrawal of water from the river, lake, or aquifer, conflicts often arise with another important objective of watershed protection: protection of freshwater biodiversity. The recent establishment of "water markets" in the western U.S., enabling the sale or lease of water rights, has created opportunities for market-based mechanisms to shift water back into rivers in order to enhance habitat for freshwater biodiversity. For example, the Environmental Resources Trust (ERT), a non-profit organization affiliated with the Environmental Defense Fund, has established an alliance with the Bonneville Power Administration (BPA) under which it has become a broker for "environmentally friendly power" produced by BPA. BPA sells four types of power that ERT defines as "environmentally friendly" because it provides benefits for fish through increased river flows (see Box 3). ERT serves as a broker for this "environmentally friendly" power and earns a fee for every unit of electricity it sells. ERT then uses 75% of those fees to support environmental restoration in the Pacific Northwest. The first sale of power under this arrangement was announced on February 19, 1998.

Box 3. Types of "environmentally friendly" power being brokered by ERT.

- *Fish flow hydropower*—power produced as a result of mandated increased discharges to improve salmon survival during downstream migration;
- *New instream flow hydropower*—power produced by the purchase or lease of water rights to increase dry year flows for salmon habitat;
- *Light-load hour hydro*—power generated during off-peak hours by water that would normally be spilled over the dams due to high spring flows. Spilled water increases the nitrogen gas concentration with harmful consequences for salmon;
- *Renewable sources*—wind, solar, geothermal.

ERT has also reached an agreement in principle with BPA under which ERT could keep all of the net revenues resulting from electricity generated by the water that stays in the river due to ERT's purchase or leases of water rights. Because more water in the river means more electricity produced from the existing hydroplants and more electricity means more revenues for the operators, the purchase of water rights generates an economic return. Thus, ERT might enter into a dry-year lease arrangement with a farmer, under which it holds rights to a portion of the farmer's water rights during particularly dry years. BPA agrees to return the net revenue generated from the additional flow to ERT, although negotiations are still underway on how to calculate the net increment in river flow and the value of the incremental power generated during discharges. ERT would then use the proceeds to purchase or lease additional water rights.

Whether motivated by the need to maintain instream flows, protect hydropower investments, or to protect urban water supplies, the presence of water markets is setting the stage for "water cropping" in the West. Farmers have always faced decisions about which crop will provide the highest economic return in any given year. Today, in parts of the western United States, farmers may well find that leasing their water rights for downstream uses provides the highest return of any of these alternatives.

All of the mechanisms explored above involve placing some mechanism for taxing or charging those who benefit from water (the public, electricity users, water users) and using those revenues to pay others to not degrade the water supply. At the same time, those mechanisms that directly link the charge to the amount of service used create an economic incentive for conserving water, lessening demand on the service, and thus increasing the effective supply for other users. One example of a policy established with water conservation as a primary goal is legislation now being considered in South Africa that would allow a tax to be established on commercial forest plantations that use alien species. Because these species consume more water than native trees, they diminish water supplies. The legislation would create an economic incentive to encourage the use of water-conserving species, while also generating revenue that could contribute to the national efforts to clear land of alien species (Preston, pers. comm.).

Scope for capturing ecosystem service values associated with water

How much scope exists to protect ecosystem services associated with water quality and quantity through mechanisms such as those described above? Most of the examples of watershed protection described above involve urban drinking water and hydroelectricity. Some cases, such as the establishment of Dumoga-Bone National park in Indonesia or Maduru Oya National Park in Sri Lanka, also involve watershed protection for irrigation, although this appears to be a less common rationale. Watershed protection for urban and hydropower uses is likely to be more common because in both cases water quality maintenance is essential. Significant

watershed development is less of a threat for agricultural water users (and indeed deforestation can often increase water discharge in some ecosystems).

To obtain a rough estimate of the scale of the opportunity that water values might provide for protecting watersheds, I examine the water needs for hydropower and urban water supply and the associated watershed area required.

1. Municipal water supply

The size of watersheds for urban water supplies vary dramatically based on the urban population being served, regional precipitation and climate, soil characteristics, characteristics of the water source (groundwater or surface water), and so forth (see Table 1). The 17,300-acre Sterling Forest, for example, is a critical re-charge zone for an aquifer providing water for some 2 million people in New Jersey. The per capita area of watershed protected in Sterling Forest is quite small, less than 1% of the per capita area protected for the surface water supply meeting New York's needs.

To estimate per capita watershed size for urban water use, I analyzed data on 74 municipal water supplies in the United States (American Water Works Association Research Foundation, 1995). For these water systems, the mean per capita watershed area is 1.55 ha per capita, and the median area per capita is only 0.30 ha (see Fig. 2). The distribution of watershed area per capita is highly skewed because some cities obtain water from large rivers (e.g., Washington, D.C., obtains water from the Potomac River) with extensive watersheds that could meet the municipal water needs of much larger populations. The median value of watershed area

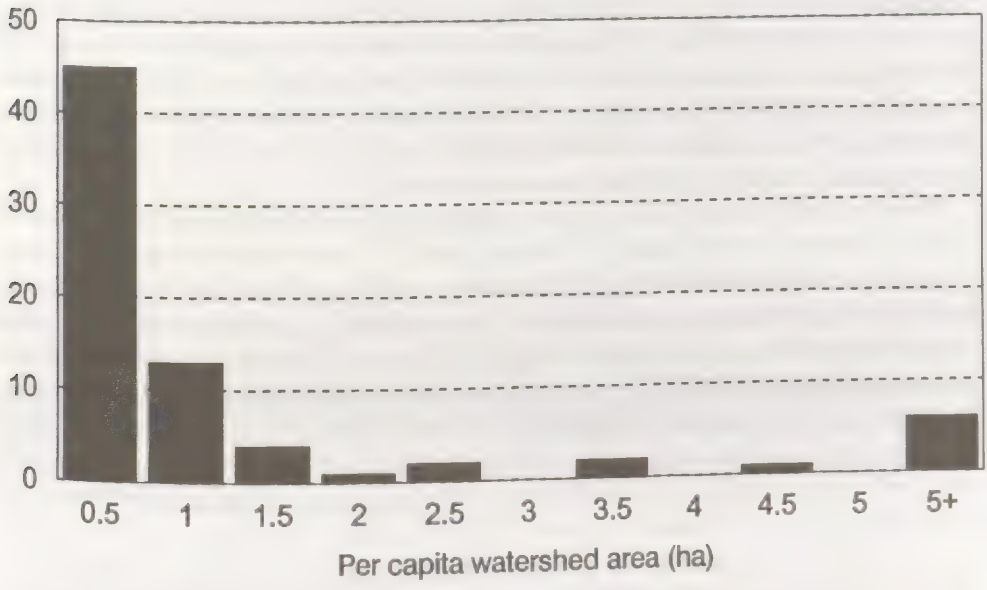


Figure 2. Frequency distribution of per capita watershed area for 74 U.S. municipal water systems.

per capita is thus more indicative of the area of land that might be managed through city actions to protect the source of its water. Cities obtaining water from large rivers are unlikely to be able to protect the source, while those relying on smaller watersheds have both the incentive and mechanisms (e.g., land purchase) to protect or carefully manage the watershed. Using the median watershed area per capita for these 74 municipalities, and focusing only on the lower 48 states, roughly 78.8 million ha serving as watersheds for municipalities could logically be protected or managed with water quality as a major goal, or 10% of land area.

Some fraction of these watersheds are already protected by federal, state, or municipal agencies. For the nation as a whole, the median percentage of watershed lands owned by water utilities is about 2%. In addition, more than 900 municipal watersheds in the U.S. are located within National Forests (Dombek, 1998). However, no statistics are available on the total fraction of municipal watershed land currently protected by municipal or other public agencies.

New regulations under the Clean Drinking Water Act have created strong incentives in the U.S. to expand the scope of watershed protection. These regulations require municipalities to install water filtration unless they have in place an approved watershed management plan. Currently, about 700 of the nation's 10,000 surface water systems are unfiltered. Of these, approximately 130 water systems legally avoid filtration (EPA Office of Ground Water and Drinking Water, unpublished data, 1998), largely through ownership of the source. The remaining unfiltered surface systems are now on various state or EPA schedules to install filtration. Nationwide, water companies will have to spend some $12.1 billion to bring their facilities into compliance with the Safe Drinking Water Act (Trust for Public Land, 1998). (For perspective, in 1996 Americans spent nearly $3 billion on bottled water and an additional $1.4 billion on home filtration systems.) Even for systems forced to install filtration or other systems for purification, watershed protection can still be a cost-effective strategy. As water quality declines, cities must invest in more expensive treatment facilities, and often the water quality is reduced in any event as levels of chlorination are increased.

What fraction of land globally might be protected for water quality based on the above estimates? Extrapolating U.S. data globally obviously makes rough estimates even rougher, but because the U.S. does span a wide range of climatic conditions, the result is still likely to be indicative of the order of magnitude of land area that might be involved for urban water supply. If we use the median watershed area per capita for the U.S. as a rough estimate of the potential watershed area required for urban use globally, current populations would require 1.7 billion ha, or 13% of land area. If population size grows to 10 billion (within the range of U.N. populations projections for the year 2050), and the per capita watershed area necessary for urban water supply remains constant, the watershed area required would grow to some 23% of land area. However, it would be even less likely at that time for the land to be managed with water quality as a primary goal since the potential economic benefits of other uses would grow with growing populations.

2. Hydropower catchment protection

How much watershed area could be protected through revenues captured from existing or planned hydropower projects? It is difficult to make any global estimates, since both the geomorphological and biological attributes of a particular dam site and watershed strongly influence the extent to which different watershed management practices might affect the longevity of a hydropower facility (Chomitz & Kumari, 1998). Even so, several examples provide a sense of scale for different countries:

Bhutan. Bhutan has significant hydropower potential, with a capacity estimated at over 20,000 MW, of which only 344 MW is currently developed (see Box 4). The country's Power System Master Plan identifies 91 potential hydroelectric sites and estimates the economically exploitable hydro resources to be more than 16,000 MW. The country sells the bulk of its electricity to India. The 1993 market price for electricity in India was 10.5 cents/KWh (Peak), 7 cents (firm), and 3.5 cents (seasonal). The cost of hydropower production in Bhutan was roughly 2.3 cents/KWh. Given India's growing demand and lack of alternative inexpensive energy sources, Bhutan will be able to develop a significant fraction of its hydropower resources and build forest conservation costs into the price of electricity. It is unlikely that Bhutan will fully develop its hydropower potential because of the environmental and social costs involved and because of economically attractive alternative uses of rivers (e.g., for ecotourism). Nonetheless, significant additional hydro development is likely to take place and, since much of the planned development is run-of-the-river, the environmental impacts are likely to be lower than is often the case with hydropower plant construction. Given the region's topography (steeply sloping from the north to south), most of the 47,000 km^2 nation is, in effect, the catchment for existing and planned hydro projects. The protection of forested lands will be critical for the maintenance of hydropower production. Since the country is 72% forested, at least this fraction of the nation's land could logically be managed with catchment protection as one goal, and with management costs financed through hydropower sales.

Venezuela. The 95,000 km^2 Caroni river basin is the catchment for 95% of Venezuela's present and planned hydro development. National hydropower capacity in 1992 was about 10,000 MW, but planned development will total some 26,800 MW. Protecting this entire basin for its hydropower potential would amount to 10.8% of Venezuela's land area of 882,050 km^2. Currently, the 3-million-ha Canaima National Park protects approximately 30% of the watershed, and its size and location were justified in part for this purpose.

Costa Rica. The Arenal Hydroelectric Project in Costa Rica has an installed capacity of 362 MW, accounting for 33% of the country's total installed capacity. The Arenal Watershed has a total land area of 41,334 ha (about 1% of Costa Rica's area),

Box 4. Hydropower development in Bhutan.

Bhutan currently negotiates the cost of hydropower sales to its primary customer, India, based on the construction and operating costs of the hydroelectric plants. The country does not include the costs of watershed protection as a portion of those operating costs. An example of a more appropriate economic arrangement can be applied to the planned Kholong Chhu hydro project. This hydro project, if built, would have an installed capacity of 290 MW and the continuity of its power production would rely heavily on the maintenance of the integrity of the upstream catchment area (an estimated 1134 km^2, or about 2.5% of Bhutan's land area).

The economic evaluation of a project like this hinges on the cost of construction, the cost of operation and maintenance, and the price that the electricity will be sold. Electricity can be produced through Bhutan's hydropower projects at a much lower cost than the current market price in India. Market prices for energy in India in 1993 were 10.5¢/kWh (peak), 7¢/kWh (firm), and 3.5¢/kWh (seasonal)—these form the "ceiling" price for electricity sales from Bhutan. So long as Bhutan can cover the capital, financing, and operation and maintenance (O&M) costs at prices lower than these, then it stands to make a profit. (The pre-feasibility study estimated the unit cost of supply to be 2.34¢/kWh; the 1993 tariff for domestic sales in Bhutan was 1.3¢/kWh.) Typically, the actual electricity costs are set through negotiation with India at a level between the cost of production and the "ceiling" price (and because India has financed past hydro development, it has additional leverage on keeping the costs low).

In the pre-feasibility study, capital expenditures for the project were estimated to be $215 million with operation and maintenance costs (O&M) estimated to be about $2 million per year for the 25-year life of the plant. The capital expenditure for the site, however, does not include the opportunity cost of other uses of the catchment area (e.g., for timber harvesting) and the O&M costs do not include the cost of protecting and sustainably managing the forests in the catchment area. In fact, a substantial fraction of these costs could logically be covered as part of the O&M costs for various hydropower developments.

For the sake of argument, Bhutan could consider that the cost of maintaining the watershed is equal to a fraction of its Forest Service Division's budget proportional to the area encompassed by the watershed. With an annual budget of about $6 million and the catchment covering about 2.5% of the country, a rough allocation of the Forest Services Budget to the O&M costs for the project would be $150,000 per year—an increase of only 7.5% in the O&M costs for the project. The net effect would be to slightly increase the O&M costs of the projects, thereby increasing the unit cost of electricity supply. If the price is negotiated from this higher base, Bhutan would then obtain greater revenues from the electricity sales, and a portion of those revenues could then be returned to support watershed protection.

Source: Reid (1996).

of which 19,108 ha is in the upper watershed. More than 90% of the upper watershed has slopes greater than 25% and thus plays an important role in erosion, streamflow, and sedimentation (Aylward et al., 1994). Management of this entire watershed for protection of its hydro potential using revenues generated from hydropower would be justified, although recent studies have shown that the most economically effective management regime from the standpoint of hydropower production does not require maintenance of significant forest cover. Loss of forest does slightly increase sedimentation, but this is more than offset by increased water yield (Aylward et al., 1998).

Nepal. The Kulekhani Hydroelectric dam produces 60 MW of electricity from a watershed of 125 km^2. The watershed is being heavily used to produce food, fodder, fuelwood, fiber, and shelter. Due to the rapid sedimentation that has resulted, the volume of the reservoir has been reduced by 66% in the past 13 years, and the planned 50-year life of the dam has been reduced to just 20 years (McNeely, 1998).

Lao People's Democratic Republic. The Lao PDR has significant hydropower resources and is located close to some of the most rapidly growing countries in the world, including Thailand and Vietnam. Some 60 hydropower projects are now under consideration in the country. The Lao PDR will export some 3000 MW of hydropower energy to Thailand alone from new capacity expected to be on line in the year 2006. The development of these hydropower resources poses significant risks for biodiversity in the country. Some 16 of the 29 projects planned for completion by 2010 present direct threats to existing protected areas because they are located inside, partly inside, or close to and upstream of protected areas. The generating capacity and watershed area for different river basins in Laos is shown in Table 2. Overall, hydropower development in Laos could justify watershed protection of some 97,303 km^2, or 42% of the nation's 230,800 km^2 total area. The Lao PDR is conducting environmental reviews for each project and putting in place Environmental and Social Action Plans to address the problems identified. For example, environmental plans for the proposed 680 MW Nam Theun 2 hydroelectric project call for expenditure of $1 million per year over 30 years for protection and management of a protected forest area that constitutes 75% of the 4000 km^2 catchment area (MIH, 1997). The estimated gross revenues to the Lao PDR rise from $10 million annually to over $100 million annually (in 1994 dollars) during the 25 years of the concession for the construction and operation of the facility, which is being developed as a build-own-operate-transfer project with Lao PDR holding 25% share and a consortium of private companies holding the remainder. The $1 million spent on catchment protection per year is thus only approximately 1 to 2% of expected gross revenues.

Table 2. Lao PDR hydropower development.

River Basin	Projects	Capacity (MW)	Basin Size (km^2)	ha/MW
Sebangnouan	1	50	1,363	
Nam Ou	2	952	20,220	
Nam Ngiep	2	935	4,510	
Nam Khan	3	355	4,940	
Sebanghieng	4	500	19,600	
Nam Theun	7	2,635	5,150	
Nam Ngum	11	1,515	13,520	
Sekong	20	2,401	28,000	
Total		9,343	97,303	1,041

Source: Berkmüller (1995).

Table 3. Catchment area per MW of installed hydropower capacity.

Country	Installed Capacity (MW)	Catchment (ha)	ha/MW	Notes
Bhutan	1,650	4,700,000	2050	Planned by 2002
	16,000	4,700,000	211	Economically exploitable
Venezuela–Caroni	26,800	9,500,000	117–373	Planned capacity. Range: entire watershed vs. only Canaima protected
Costa Rica–Arenal	362	41,334	53–114	Range: upper watershed vs. entire watershed
Nepal–Kulekhani	60	12,500	208	
Lao PDR	9,343	9,730,000	1041	Proposed development

At the beginning of the 1990s, the total world installed hydroelectricity capacity was 615,000 MW, with North America and Europe having developed 60 and 36% of their hydropower potentials, respectively, Asia and Latin America about 10%, and Africa only 5% (Gleick, 1994). A large fraction of the remaining hydropower potential is in tropical forest areas—in South America, as much as 80% may be in tropical forest regions (Moreira & Poole, 1993). Clearly, the amount of watershed that might be protected per unit of electricity-generating potential will vary widely depending upon rainfall and topography. In the examples cited above, one project in relatively steep Nepal involved as little as 208 ha of watershed per MW, while the extensive development planned for Lao PDR would involve watershed areas of 1041 ha per MW (Table 3). Using these five examples, and choosing values for the maximum proposed hydropower development and the entire watershed area, an average of about 400 ha of watershed could be managed with catchment protection as a major objective per MW of installed capacity. Given current worldwide installed capacity, this would amount to 246 million ha (2% of land area) and with economically exploitable hydro potential estimated to be 3 to 4.5 times current levels of production, the watershed area involved would be 738 million ha to 1107 million ha or 6 to 8% of total land area.

Future prospects for capturing watershed values

These are at best ballpark estimates of the area that might be managed for ecosystem service values based on revenues captured from those services for hydropower and drinking water. It is certain, though, that the incentive for managing watersheds for these water values will rise in coming years for three main reasons.

First, growing water scarcity will increase the market value of water. The amount of freshwater withdrawn from rivers and groundwater has risen 35-fold in the past 300 years, with a 4-fold increase between 1940 and 1990 (Shiklomanov, 1993). As this demand for water has been rising, the supply of clean water has been shrinking due to contamination by pollutants. On the basis of past experience, renewable freshwater resources of 1000 cubic meters per person per year defines a

reasonable threshold for "water stress," below which countries are likely to experience water scarcity sufficient to impede development and harm human health. In 1990, some 20 countries, with a population of about 132 million people, fall into this water stress category. By the year 2025, this figure is projected to grow to 653–904 million people, and by 2050 could reach as high as 2.4 billion people, representing 20% of the projected world population (World Resources Institute, 1996).

Second, increased recognition of the public health gains that can be achieved through access to clean water has stimulated major investments in efforts to improve water quality. During the 1980s, nearly $100 billion in investments were made as parts of the International Drinking Water and Sanitation Decade, during which time the number of urban residents in developing countries with access to adequate water increased by nearly 80%. Even so, in 1994, 13% of the urban population of developing countries still lacked access to safe and reliable water sources (World Resources Institute, 1996). The U.S. EPA recently estimated that treating, storing, and delivering safe drinking water to the U.S. alone will require a capital investment of $138.4 billion over the next 20 years (Trust for Public Land, 1997). These investments in clean water are often well justified, but they do raise the question of how to best target the investment. Shanghai, China, for example, moved its water intakes 40 km upstream at a cost of $300 million because of the degradation of water quality near the city. In other regions, investment in protection of the water quality may be a more cost-effective solution than new infrastructure or water treatment facilities.

Third, demand is growing for water for in-stream uses, and in particular for the protection of biodiversity and biological resources in freshwater ecosystems. Freshwater biodiversity is by far the most threatened of all of the world's biodiversity. In North America, 65% of crayfish, 37% of fish, and 38% of amphibians are either in jeopardy of extinction or extinct (The Nature Conservancy, 1996). Worldwide, an estimated 34% of freshwater fish species are vulnerable, endangered, or critically endangered (IUCN, 1996). As countries act to conserve biodiversity, the legal and public pressure to maintain adequate instream flows will grow.

II. Carbon Sequestration

Unlike ecosystem services involving water, there has been relatively little experience with mechanisms for capturing value associated with carbon sequestration as a means of financing the protection of that service. Several studies have explored the role that forest and land-use activities may play as a source of emissions of CO_2 to the atmosphere or, alternatively, as a mechanism for slowing the build-up of this greenhouse gas in the atmosphere. But as yet, no formal mechanism for capturing values associated with carbon emissions have been established, although some experimental efforts have begun under the United Nations Framework Convention on Climate Change (UNFCCC).

Mechanisms for capturing ecosystem service values related to carbon

The U.N. Framework Convention on Climate Change, which came into force in 1993, did not include any specific mechanism under which countries could obtain credit for steps to control greenhouse gas emissions through land-use changes. Countries could, however, logically include actions to sequester carbon through land-use changes as contributions to achieving the voluntary commitments made by Annex 1 countries (primarily OECD countries and the Former Soviet Union). In addition, the UNFCCC allowed experimentation with "joint implementation," a mechanism where a country or company could pay for emission reductions in another country. These so-called "Activities Implemented Jointly" included various land-use and forest projects designed to sequester carbon.

The Kyoto Protocol, agreed to on 11 December 1997, has taken significant steps forward in establishing mechanisms under which a country can gain economic benefits by reducing land-use carbon dioxide emissions. Specifically, among Annex 1 countries, net changes in removals or emissions of carbon dioxide related to afforestation, reforestation, and deforestation will count toward the emission caps that they negotiated. In addition, Annex 1 countries can meet their reductions through carbon mitigation projects undertaken in other Annex 1 countries, and these projects can include enhancement of carbon sinks (through, for example, afforestation) or reduction of emissions (through, for example, slowing of forest conversion). Finally, the Kyoto Protocol establishes the "Clean Development Mechanism" (CDM), under which Annex 1 countries can meet their reductions through projects in non-Annex 1 countries. Although the details of the CDM remain to be defined, this mechanism could provide a means by which a developed country could pay a developing country to afforest areas that would not otherwise have been forested and count some portion of the carbon sequestered against the developed country's targets.

Just how these various trading mechanisms will ultimately function remains to be seen. Indeed, it is still an open question whether forest and land-use projects will be included in certain mechanisms (such as the CDM) and, if they are included, it is not yet clear which types of management practices will be allowed. Already, the language of the Kyoto Protocol may exclude the protection of standing forests from counting toward certain targets. (Deforestation, for example, can count as a source of emissions, but it is not yet clear whether countries will be able to claim carbon credits in the CDM for actions that result in the avoidance of deforestation.)

Demonstration projects allowed during the pilot phase of Joint Implementation under the FCCC, have shown the potential for land-use carbon offsets as a means to mitigate climate change. A number of governments and companies (particularly electric utilities) have invested in land-use carbon sequestration activities either as voluntary offsets for their own carbon emissions (e.g., Applied Energy Services, a U.S. utility), or as a means of evaluating the potential for carbon offsets. Costa Rica, which has aggressively pursued carbon sequestration as a means of financing forest conservation in the country, has established a Greenhouse Gas Fund to pro-

mote Joint Implementation projects. Investors seeking to offset carbon emissions contribute to the fund and receive in return "Certifiable Tradable Offsets" (CTOs), based on a bundle of nationally certified sequestration projects. The government anticipates that these CTOs may ultimately be used as credits against greenhouse gas emissions.

What is the scope for conservation related to carbon services?

The greatest potential for slowing carbon emissions or for sequestering carbon through land-use practices lies in the tropics where current rates of deforestation are high. In contrast, temperate forest area has been relatively stable and is actually increasing in some regions. The most detailed study of the potential scope of tropical-forest-related projects for mitigating climate change concluded that while some 650 million ha of tropical forest are likely to be lost by 2050 under a "business as usual" projection, this loss could be reduced by about 138 million ha through policies and economic incentives associated with carbon sequestration. An additional 220 million ha of degraded forest land could be regenerated during this time period, and some 128 million ha could be planted through farm forestry and plantations (Trexler & Haugen, 1995). Given a total tropical forest area of 2727 million ha (World Resources Institute, 1996), the potential deforestation that could be avoided amounts to 5% of tropical forests (about .3% of earth's land surface). Including the potential for regenerating forests (but excluding farm forestry and plantations, which would play relatively little role in biodiversity conservation), the scope for land-use carbon offsets in the tropics grows to 13% of tropical forests (or, about 1% of land surface).

Assuming that forest protection and reforestation are ultimately included in trading mechanisms under the Climate Convention, how much of this 5–13% of tropical forest area would actually be economically attractive for carbon mitigation? Potentially, much or all of this area would provide lower-cost carbon offsets than alternatives. The cost of land-use carbon sequestration projects that have been implemented over the past decade ranges from about 37 cents (Bolivia) to $6 per ton (Costa Rica) of carbon, well below the costs projected for controlling industrial emissions of carbon dioxide. Trexler and Meganck (1993) estimated that under a fully functional mechanism for land-use carbon sequestration, demand would increase the price to $5 to $10 per ton, and with a potential annual sequestration of 1 billion tons of carbon in tropical forests the potential exists for $5 to 10 billion in financial flows for land-use projects in developing countries each year. (By way of comparison, 1995 global Official Development Assistance to all developing countries totaled $59.8 billion.)

III. Bio-Control and Pollination Services

According to Costanza et al. (1997), the economic value provided by ecosystems in the form of natural pest control and pollination are relatively small, representing only 1.6% of the total value of ecosystem services, compared to such services as

nutrient cycling (51% of the value of ecosystem services) and cultural values (9% of the value). Unlike many other ecosystem services, however, some fraction of the economic values of biocontrol or pollination could, in theory, be captured by those benefiting economically from the services and used to finance their conservation. (Indeed, traditional farming systems, which use little or no chemical inputs, rely entirely on the benefits of these ecosystem services.) Farmers who lose the benefit of the pollinator or pest control services provided by adjacent wildlands must pay for a replacement of those services in the form of pesticides and commercial pollination services. Thus, they would have an incentive to pay the cost of protecting the service if it is cheaper than substitutes for the service.

In practice, apart from traditional agricultural systems, there are few examples of an individual or institution creating a mechanism to capture ecosystem service

Box 5. Bundling Ecosystem Services: Del Oro.

One of the most innovative examples of a mechanism to capture the value of ecosystem services is an agreement pending between the Guanacaste Conservation Area (GCA) in Costa Rica and Del Oro S.A., an orange growing corporation. The GCA covers 120,000 ha on land and an additional 70,000 marine ha, and contains an estimated 65% of Costa Rica's plant and animal species. The Conservation Area provides significant economic benefits through ecotourism, local employment, and, more recently, as the source for chemicals of interest to the pharmaceutical and agrochemical industry. The GCA also, however, has protected a number of ecosystem services of value to surrounding lands, which until now have been provided freely. In particular, the GCA protects important watersheds and harbors natural pest control species. One of the beneficiaries of these services is the Del Oro Company.

Although Del Oro's land is adjacent to the GCA, not all of the land was suitable for orange production. Some 1200 ha of its land remained forested and could, potentially, have been sold or developed for alternative uses. From an ecological standpoint, this forested land is extremely important, representing some of the last remaining forest in the transition zone between the hardwood forests of Guanacaste and the humid forest of the Caribbean lowlands. If this forested area, which occurred in the form of wide peninsulas in the area of the Del Oro plantation, were deforested, however, much of the value that it provided the orange groves would be lost. Moreover, the plantations also obtained significant benefits from the adjacent GCA itself. Under the agreement that has been reached, Del Oro will "pay" the GCA in the form of the 1200 ha of land in exchange for the value of the services provided by this land and the larger GCA, and as part of the GCA that Conservation Area will assume responsibility for protecting the forest. More specifically, under the 20-year contract, Del Oro agrees to pay the GCA $1/ha per year for pest control supplied by the GCA's insects for the 1685 ha of orange plantations, $5/ha per year for water supply and watershed protection for the portion of the catchment area of the river serving as the plantation's water source that is within the GCA (1169 ha), and $12 per truckload of waste orange peels for degrading the organic material at a site that the GCA would develop. The GCA also would charge for the establishment of an international council and resident advisor (minimum of 13 days per year), and for the use of 1 ha of land within the GCA to maintain oranges free from potential disease that might inflict the plantation.

The total value for these services amounts to $24,000 per year, or $480,000 for the 20-year contract. In lieu of that payment, Del Oro will donate the 1200 ha of forested land to the GCA.

Source: Dan Janzen (U. Penn., pers. comm.).

values related to pest control or pollination. One of those examples is the recent agreement between the Guanacaste Conservation Area (GCA) in Costa Rica and the adjacent Del Oro Company, under which the GCA will be paid $1 per ha per year for supplying natural pest control services to the 1685-ha orange plantation that the company established adjacent to the conservation area (see Box 5).

The use of hedgerows in agricultural systems in Germany has also been attributed, in part, to the biocontrol services that they provide. Hedgerows provide habitat for herbivorous insects and aphids, which in turn support a large number of predators and parasites in their natural setting. When the grain crop is developing, natural enemies supported in the hedgerows control the potential aphid damage and, as a result, pesticides are not needed to control aphids in northeast Bavaria. Hedgerows, however, are becoming less common in this region. The value of natural pest control is insufficient to offset the economic benefits associated with larger fields and the use of mechanized equipment, both of which result in less "edge" suitable for hedgerows (Naylor & Ehrlich, 1997). (This shift to greater use of pesticides and mechanized equipment, however, may be stimulated in part by the economically inefficient subsidies to Europe's agriculture sector.)

Within the United States, the protection of natural habitat for pollinator or pest control services is rare. Agricultural land left uncultivated for other reasons, for example, due to steep slope or soil conditions, is now sometimes managed by farmers for purposes including biocontrol. Further, certain regulations now require land to be left fallow. For example, in order to be certified as "organic," organic producers are required to leave unplanted buffer strips around their fields to lessen the likelihood of pesticide residues and these buffers, in turn, may be helpful from the standpoint of biotic control. But it is rare to find land left fallow specifically to serve as habitat for biocontrol agents.

The capture of ecosystem values related to pollinator services or biotic control does not appear to be economically justified. It is difficult to design mechanisms that circumvent the public good attributes of pollinator and pest control services. A farmer who leaves land unplanted pays a significant opportunity cost and captures only a portion of the benefits of the services. Farmers could protect land through cooperative approaches, but it would be difficult to determine the appropriate distribution of protected land in relation to the crops requiring the services. And, most important, it appears that the opportunity cost of leaving land unplanted typically is far greater than the economic benefits obtained from natural pest control. Nevertheless, consumer preferences combined with the removal of subsidies and policies that favor high-input industrial agriculture could ultimately make biocontrol and pollinator services more economically attractive.

IV. Waste Treatment

Wetlands have long been known to be cost-effective means for wastewater treatment. Requirements for tertiary treatment of wastewater in the United States greatly

elevated the costs of wastewater treatment facilities, with tertiary treatment often doubling the cost of existing primary and secondary treatment facilities. Wetlands are often capable of providing this tertiary treatment at a fraction of the cost of a new facility and, as a consequence, several hundred wetlands are used in this manner in the United States and Canada. However, wastewater does alter the composition and functioning of natural wetlands and as a consequence, interest in using natural wetlands for this purpose has declined as recognition of the importance of wetlands for other values such as biodiversity conservation has grown.

Artificial wetlands still often provide economically attractive alternatives to tertiary treatment facilities, particularly for relatively small communities. They have also been used for wastewater treatment for some larger cities in developing countries, such as Calcutta. While there is no doubt some scope for expansion of the use of artificial wetlands for wastewater treatment, the fact that they have not proven suitable for larger urban areas in industrialized countries suggests that they will not become a significant type of land use in these regions, although they might be more significant in developing countries. In any event, artificial wetlands being used for wastewater treatment would not play a significant role in biodiversity conservation.

In tropical regions, mangroves have long been recognized to play a similar role in water purification. Lal (1990), for example, calculated that the cost of treating wastewater now treated by Fiji's mangroves with a conventional treatment facility would be approximately $5820/ha/yr. But despite the clear economic value of mangroves for waste treatment, there are, as yet, few examples of situations where mangroves have been protected because of the ecosystem service that they provide or where this protection is financed through values associated with that service. One of the few examples where there appears to be a tie between mangrove conservation and the service of waste treatment is, surprisingly, shrimp aquaculture. Shrimp aquaculture was initially a major threat to mangrove forests, because mangrove ecosystems were often converted to aquaculture sites. More recently, however, shrimp aquaculture projects are being developed inland from mangrove forests because the effluent from the aquaculture system can then be purified by the mangroves. The success of the aquaculture system is dependent on maintaining high coastal water quality (The Economist, Anonymous, 1998).

V. Storm and Flood Protection

Wetlands, mangroves, and coral reefs often provide human settlements with significant protection from storms and floods. Typically, there is substantial cost involved in replacing these services with canals, coastal barriers, or dikes if the habitat is degraded or destroyed. The U.S. Army Corps of Engineers, for example, advocated the retention of a wetlands complex along the Charles River outside of Boston, Massachusetts, because of the flood control service that it provided. Under the Charles River plan, some 8500 acres of wetlands were to be acquired for the

purpose of preserving their natural flood storage. The alternative, channelizing a 10-mile stretch of the river, would cost some $30 million (Interagency Task Force, 1978).

At a smaller scale, grape growers in Napa Valley, California, are allowing river banks to return to natural vegetation in order to reduce flood damage to their crops. Grape growers traditionally planted right up to the river bank, which destabilized the bank and increased the likelihood that floodwaters would break through the natural levees. The Napa Resource Conservation District developed a model allowing assessment of the optimal buffer strip for individual farms. The buffer takes land out of production, but reduces the likelihood that floods will damage the entire grape crop. Farmers in the region now often leave buffers of 10 or more feet along river banks (D. Bowker, Resource Cons. District, pers. comm.).

There is no doubt that protection of ecosystems for the storm or flood protection that they provide is often economically justified. There are no end of problems, however, in establishing mechanisms for capturing those values and using them to protect the service. Consider, for example, the wetlands in the upper Mississippi River drainage that protect downstream cities from flooding. The extent to which the wetlands perform this function varies from year to year depending upon the timing and severity of rainfall and depending on the location of the cities involved. A national program to protect those wetlands thus makes sense, but the design of any state or local programs that might tax residents for wetland conservation would be daunting. It is not clear that those municipalities would pay the replacement cost of the service to begin with.

Conclusion

Knowing the economic value provided by ecosystem services can help us appreciate our dependence upon living systems and establish the basis for public policies to protect those services. But the fact that ecosystem services have economic value does not, in itself, mean that action to protect those services is warranted. Oxygen production, nutrient cycling, and soil formation, for example, are services that have economic value but are neither scarce nor threatened and are not in need of protection. For a number of services, however, human actions are degrading the service and this is resulting in economic costs to society, either in terms of the lost service itself or as a result of the cost of replacing the service. The services of this nature addressed in this paper include water quality and quantity, carbon sequestration, biotic control, pollination, flood and storm protection, and waste control.

Two general approaches can be used to maintain these services. First, the public sector can take action to protect the services by using general revenues or eminent domain to establish protected areas or regulate land development. Such public actions are often justified as well by virtue of the non-commercial values that might be protected through such action, such as the protection of those attributes of biological diversity that may have little or no economic value. But the scope for pub-

lic action is limited by society's many competing demands. Alternatively, the service can be protected if mechanisms can be put into place that can capture some of the economic value associated with the service and use this to finance its protection.

What is the potential for protecting ecosystem services, and the biodiversity associated with them, through mechanisms that more directly capture the value of the service? For ecosystem services associated with water quantity and quality a significant scope for action exists. Management of some 13% of land area for its watershed value for municipal water supplies could currently be justified, and this could grow to 23% by 2050. Some 2–8% could be managed primarily to maintain the value of hydropower plants relying on those watersheds. In the case of carbon sequestration, the potential is again significant. As much as 5–13% of tropical forests could be managed with carbon sequestration as a major goal, but as yet, no mechanism is in place to facilitate this. The design of the Clean Development Mechanism under the Climate Convention will determine whether this potential can be achieved.

For a number of other ecosystem services, including pollination, pest control, waste treatment, and flood and storm protection, opportunities do exist to capture some portion of the value associated with these services. However, in general, the scope for using such mechanisms as a source of direct financial support for the conservation of the service appears to be limited.

Acknowledgments

I would like to thank Theresa Bradley, Geoffrey Heal, and Nels Johnson, Jeffrey McNeely, and one anonymous reviewer for comments on earlier drafts of this manuscript and Alexandra Sevilla for assistance with research and manuscript preparation.

Literature Cited

American Water Works Association Research Foundation. 1995. Electronic Watershed Management Reference Manual Version 1.0. American Water Works Association Research Foundation, Denver, Colorado.

Anonymous. 1998. Science and technology: Going swimmingly. The Economist 346(8056): 80–81.

Aylward, B., J. Echeverría & E. B. Barbier. 1994. Economic Incentives for Watershed Protection: A Report on an Ongoing Study of Arenal, Costa Rica. CREED Working Paper Series No. 3, International Institute for Environment and Development, London, U.K.

Aylward, B., J. Echeverría, A. Fernández González, I. Porras, K. Allen & R. Mejías. 1998. Economic Incentives for Watershed Protection: A Case Study of Lake Arenal, Costa Rica. Final Report to the Government of the Netherlands under the program of the

Collaborative Research in the Economics of Environment and Development (CREED). International Institute for Environment and Development, London, U.K.

Berkmüller, K. 1995. Hydropower development and protected areas. Unpublished manuscript, IUCN, Gland.

Budrock, H. 1997. Summary Guide to the Terms of the Watershed Agreement. The Catskill Center for Conservation and Development, Arkville, New York.

Chichilnisky, G. & G. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–630.

Chomitz, K. M., E. Brenes & L. Constantino. 1998. Financing environmental services: The Costa Rican experience and its implications. Unpublished manuscript, Development Research Group, World Bank, Washington, D.C.

Chomitz, K. M. & K. Kumari. 1998. The domestic benefits of tropical forests: A critical review. The World Bank Research Observer 13(1): 13–35.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Daily, G. C. (Editor). 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Department of Water Affairs and Forestry. 1997. The Working for Water Program: 1996/97 Annual Report. Ministry of Water Affairs and Forestry, South Africa.

Dombek, M. 1998. A gradual unfolding of national purpose: A natural resource agenda for the 21st Century. Speech given in Bozeman, Montana, March 2, 1998.

Garcia, J. R. 1984. Waterfalls, hydropower, and water for industry: Contributions from Canaima National Park, Venezuela. Pp. 588–591 *in* J. A. McNeely & K. R. Miller (editors), National Parks, Conservation, and Development: The Role of Protected Areas in Sustaining Society. Smithsonian Institution Press, Washington, D.C.

Gleick, P. H. 1994. Water and energy. Annual Review of Energy and Environment 19: 267–299.

Gutman, P. Unpublished ms. Putting a price tag to biodiversity: Cost-benefit analysis of national parks conservation in Venezuela. Center for Urban and Regional Studies, Buenos Aires, Argentina, December 1995.

Interagency Task Force. 1978. Our Nation's Wetlands. Council on Environmental Quality, Washington, D.C.

IUCN. 1996. 1996 IUCN Red List of Threatened Animals. World Conservation Union (IUCN), Gland, Switzerland.

Lal, P. N. 1990. Conservation or Conversion of Mangroves in Fiji. Occasional Papers of the East-West Environment and Policy Institute. Paper No. 11, East West Environment and Policy Institute, Honolulu, Hawaii.

Le Maitre, D. C., B. W. Van Wilgen, R. A. Chapman & D. H. McKelly. 1996. Invasive plants and water resources in the Western Cape Province, South Africa: Modeling the consequences of a lack of management. Journal of Applied Ecology 33: 161–172.

McNeely, J. A. 1998. Mobilizing Broader Support for Asia's Biodiversity: How Civil Society can Contribute to Protected Area Management. Asian Development Bank, Manilla, Philippines.

McNeely, J. A., K. R. Miller, W. V. Reid, R. A. Mittermeier & T. B. Werner. 1990. Conserving the World's Biodiversity. World Resources Institute and IUCN, Washington, D.C., and Gland, Switzerland.

MIH [Ministry of Industry and Handicraft]. 1997. Nam Theun 2: Study of Alternatives. MIH, Hydropower Office, Vientiane, Lao PDR.

Moreira, J. R. & A. D. Poole. 1993. Hydropower and its constraints. Pp. 73–119 *in* T. B. Johansson, H. Kelly, A. K. N. Reddy & R. H. Williams (editors), Renewable Energy: Sources for Fuels and Electricity. Island Press, Washington, D.C.

Myers, N. 1984. The Primary Source: Tropical Forests and our Future. W.W. Norton, New York.

Myers, P. 1997a. Voters go Green. GreenSense 3(1): 3. The Trust for Public Land. San Francisco, California.

Myers, P. (Editor). 1997b. Ways and Means. GreenSense 3(2): 4. The Trust for Public Land. San Francisco, California.

Nature Conservancy, The. 1996. Priorities for Conservation: 1996 Annual Report Card for U.S. Plant and Animal Species. The Nature Conservancy, Arlington, Virginia.

Naylor, R. L. & P. R. Ehrlich. 1997. Natural pest control services and agriculture. Pp. 151–174 *in* G. C. Daily (editor), Nature's Services. Island Press, Washington, D.C.

Pearce, D. 1998. Auditing the earth. The value of the world's ecosystem services and natural capital. Environment 40(2): 23–28.

Pimentel, D., C. Wilson, C. McCullum, R. Huang, et al. 1997. Economic and environmental benefits of biodiversity. BioScience 47(11): 747–757.

Portland Water Bureau, 1997. Unpublished information sheet. Portland Water Bureau, Portland Oregon.

Revkin, A. 1997. New York begins spending to save City's reservoirs. New York Times, January 22, 1997: A1.

Reid, W. V. 1996. Biodiversity Policy Options for Bhutan. National Environment Committee, Royal Government of Bhutan, Thimphu, Bhutan. Unpublished manuscript.

Reid, W. V., S. A. Laird, C. A. Meyer, R. Gamez, A. Sittenfeld, D. H. Janzen, M. A. Gollin, C. Juma (editors). 1993. Biodiversity Prospecting: Using Genetic Resources for Sustainable Development. World Resources Institute, Washington, D.C.

Royal Government of Bhutan. 1996. Eighth Five Year Plan (1997–2002). Ministry of Planning, Thimphu, Bhutan.

Sagoff, M. 1996. On the value of endangered and other species. Environmental Management 20: 897–911.

Sagoff, M. 1997. Can we put a price on nature's services? Report from the Institute for Philosophy and Public Policy 17(3): 7–12. University of Maryland.

Schiklomanov, I. A. 1993. World fresh water resources. *In* P. H. Gleick (editor), Water in Crisis: A Guide to the World's Fresh Water Resources. Oxford Univ. Press, New York.

Schultz, S. D. & B. E. Lindsay. 1990. The willingness to pay for groundwater protection. Water Resources Research 26: 1869–1875.

State of New York. 1998. The New York City Watershed Agreement: Memorandum of Agreement Final Draft. Internet address: http://www.state.ny.us/watershed/overview.html.

Trust for Public Land. 1997. Protecting the Source. Trust for Public Land, San Francisco, California.

Trust for Public Land. 1998. An Ounce of Prevention: Land Conservation and the Protection of Connecticut's Water Quality. Trust for Public Land, Boston, Massachusetts.

Trexler, M. C. & C. Haugen. 1995. Keeping it Green: Tropical Forestry Opportunities for Mitigating Climate Change. World Resources Institute, Washington, D.C.

Trexler, M. C. & R. Megangk. 1993. Biotic carbon offset programs. Climate Research 3: 129–136.

Van Wilgen, B. W., R. M. Cowling & C. J. Burgers. 1996. Valuation of ecosystem services: A case study from South African fynbos ecosystems. BioScience 46(3): 184–189.

Vogel, J. H. 1997. The Successful Use of Economic Instruments to Foster Sustainable Use of Biodiversity: Six Case Studies from Latin America and the Caribbean. Biopolicy 2, paper 5. Online journal—URL: http://www.bdt.org.br/bioline/py.

World Resources Institute. 1996. World Resources: A Guide to the Global Environment. World Resources Institute, Washington, D.C.

Ecological Economics in Poor Countries: The Current State and a Program for Improvement

Partha Dasgupta
University of Cambridge and
Beijer International Institute of Ecological Economics,
Stockholm

The Resource-Basis of Human Well-Being and its Neglect

People in poor countries are for the most part agrarian and pastoral. In 1988 rural people accounted for about 65% of the population of what the World Bank classifies as low-income countries. The proportion of the total labor force in agriculture was a bit in excess of this. The share of agriculture in gross domestic product in these countries was 30%. These figures should be contrasted with those from industrial market economies, which are 6% and 2% for the latter two ratios, respectively.

Rural communities in poor countries are biomass-based subsistence economies, in that poor people eke out a living from products obtained directly from plants and animals. For example, studies in the Indian sub-continent have shown that as much as 40–50% of the working hours of villagers are devoted to fodder and fuel collection, animal care, and grazing. Moreover, inquiries in central and west Africa have revealed the importance of forest products in rural people's lives. Poor countries, especially those in the Indian sub-continent and sub-Saharan Africa, can be expected to remain largely rural economies for some while yet.

The dependence of poor countries on their natural resources, such as soil and its cover, water, forests and their products, animals, and fisheries should be self-evident: ignore the environmental resource-base, and we are bound to obtain a misleading picture of production and consumption activities there. Nevertheless, if there has been a single thread running through nearly 50 years of investigation into the poverty of poor countries, it has been a neglect of this base. Until very recently, environmental resources made but perfunctory appearances in government planning models, and were cheerfully ignored in most of what goes by the name "development economics." There were exceptions, of course; what I am

referring to here is a neglect of environmental matters in what one could call "official" development economics. Thus, for example, the 1986 World Congress of the International Economic Association, held in New Delhi, had as its theme the appropriate balance between industry and agriculture. In the 16 sessions that comprised the Congress, none was devoted to the environmental resource-basis of production and consumption.

Handbooks offer another kind of illustration, reflecting as they do the attention researchers have given to various fields of inquiry. To me it is revealing that the two-volume *Handbook of Development Economics*, edited by Hollis Chenery and T. N. Srinivasan (North Holland, Amsterdam, 1988), contains nothing on environmental matters. The environment appears in official development economics in much the same way as that most famous bark does in literature: it does not.

In a similar vein, environmental and resource economics have neglected issues concerning rural poverty in poor countries. Thus, the three-volume *Handbook of Natural Resource and Energy Economics*, edited by Allen Kneese and James Sweeney (North Holland, Amsterdam, 1985, 1993), has nothing on poverty and its possible links with environmental degradation. Development and environmental economics have until recently passed each other by, and it is interesting to speculate why.

A prime reason often aired is that environmental and resource economists in western industrial countries (especially the U.S.A.) have tended to focus on such problems as local air pollution (e.g., sulphur emissions) and deterioration of amenities (e.g., national parks, beaches, and coastlines).[1] To the development economist, environmental matters, thus, appeared a trifle precious, not wholly relevant to the urgencies of poor countries. On innumerable occasions I have had this explanation offered to me by social scientists in developing countries. I would not wish to doubt it, but the explanation does not tell us why, when they studied development problems, these same social scientists ignored their own environmental resource-base.

At an academic level, the lack of contact between development and environmental economics has been harmful to both. Specialized fields are often driven by internal logic, so the gap between topics that are most intensively discussed and those that are most urgent and at once tractable can become large. As it happens, there was an additional loss due to the unwillingness of development and environmental economists to talk to one another. The economics of the environment, heavily involved as it is with the science of ecology, is an exciting subject; it is also a beautiful subject: there is much in it to enthuse young people.

There have been exceptions, of course. Over the years, a number of social scientists have developed environmental and resource economics within the context of rural populations in poor countries. We now have in hand a body of work that studies the links between rural poverty, population growth, and the environmental resource-base. The work has involved a fusion of theoretical modeling to empiri-

[1] For confirmation, see the article by Cropper and Oates (1992) entitled "Environmental Economics: A Survey," which reports almost exclusively on valuation techniques and cost-effective regulation of pollution.

cal findings drawn from a number of disciplines; most notably anthropology, demography, ecology, economics, geography, and political science. It offers something like a new perspective on economic development.[2] In recent years, the twin subjects, *resource economics* and *environmental economics*, have on occasion been together labeled *ecological economics*. In the remainder of the article, I will follow this practice.

The foregoing remarks have stressed developments in the supply of ideas. There have also been changes in demand. As regards timing, the shift in attitude can probably be identified with the publication of the "Brundtland Report" (World Commission, 1987), and today no account of economic development would be regarded as adequate, even by economists in developing countries, if the environmental resource-base were absent from it. The World Bank's excellent *World Development Report 1992* is an affirmation of this changed viewpoint.

But this new awareness, on its own, will not be enough. Until the environmental resource-base becomes *commonplace* furniture of economic thinking and modeling in poor countries, it will continue to be neglected in the design and implementation of public policies there. And even if they are implemented, there is the danger that they will be reached through ad hoc measures. Moreover, teachers, researchers, and policy-makers will remain oblivious of the tools that are available for the analysis of social and economic problems. They will also remain unaware of the need for scientific information on the processes that govern ecological systems and their functions, and so they may not appreciate an allied point, that there is an urgent need for poor countries to encourage the development of information channels that are conducive to good governance. It can only be hoped that tomorrow's leaders in developing countries will not only be more sensitive than today's leaders to the ecological concerns facing their own citizens, but that they will also think on such concerns in an unforced and effective way.

Tomorrow's decision-makers are today's students. So, if ecological concerns are to become an essential part of economic thinking in developing countries, ecological economics has to enter university teaching there. In order for this to happen, university teachers of economics at those universities need to learn the subject. This thought has been the basis of a *Programme of Teaching and Research in Developing Countries* that has been initiated at the Beijer International Institute of Ecological Economics of the Royal Swedish Academy of Sciences, Stockholm, by the Institute's Director, Professor Karl-Göran Mäler, and myself.[3] In the following section I will describe the program (I will call it the Programme). In Section III I will draw lessons

[2] See, for example, C.S.E. (1982, 1985), Dasgupta (1982, 1993, 1995, 1996), Anderson (1987), Agarwal and Narain (1989), Barrett (1989), Dixon et al. (1989), Falconer and Arnold (1989), Bojö et al. (1990), Dixon, James and Sherman (1990), Binswanger (1991), Dasgupta and Mäler (1991, 1995, 1997), Solorzano et al. (1991), World Bank (1992), Dixon et al. (1994), and Ehrlich et al. (1995). I am stressing rural economies, not because urban environmental problems don't exist in poor countries—they are rampant—but because the rural sector reveals a novel character in the nature of environmental problems in poor countries.

[3] A precursor to the programme was directed by us at the United Nations University's World Institute for Development Economics Research (WIDER), Helsinki, in 1989 with the encouragement of WIDER'S then Director, Dr. Lal Jayawardena. A two-volume collection of essays (Dasgupta & Mäler, 1997) that attempt to codify key problems in the area was one output of the programme.

from it, and in Section IV I will suggest what the lessons seem to indicate as regards future capacity-building in ecological economics in developing countries.

The Beijer Teaching and Research Programme in Developing Countries

The Programme has been designed to reach university teachers of economics. Its first component consists of *teaching workshops* for those among our young colleagues there who have expressed an interest in teaching and research in ecological economics. At each workshop, some 20 participants from developing countries attend a specially designed course on ecological economics. In order to save on travel cost, the workshops are held regionally, each held over a period of 10–12 days. Participants and Programme staff live and eat together in the same hotel complex.[4] This means that intellectual and social interactions occur from breakfast through to late evening every day. This interaction has been vital to the Programme. All expenses are borne by the Beijer Institute. It is our hope that over the next two years or so, some 150 university teachers of economics in developing countries will have attended them.

Given that there are no suitable textbooks on ecological economics for use in developing countries, we have had to develop the syllabus ourselves and improve upon it as we have gained experience. This has been possible only because a group of our colleagues have joined us to form something like a "permanent" teaching staff.[5] The reading list (and the lecture course) includes articles and book chapters that offer, among other things, (i) basic ecology and (ii) advanced economic analysis. We spend about three days on (i) and (ii) *before* entering the meat of the subject, namely, ecological economics in the development context (see Sections III and IV).

To date, there have been six teaching workshops: one in Sri Lanka (for participants from sub-Saharan Africa and the Indian sub-continent), one in Malta (for participants from sub-Saharan Africa), one in Malaysia (for participants from South-east Asia and Bangladesh), one in Jamaica (for participants from the Carribbean and Central America), one in Chile (for participants from South America), and one in South Africa (for participants from sub-Saharan Africa). In short, the network so created now includes about 120 university teachers from the South (I am using the term in its current geo-economic sense). I stress that the Programme could not have been launched and sustained but for collegiate friendship among the teaching staff: there are little-to-no financial rewards!

Teaching workshops, on their own, are not enough. As a minimum, they need to be supplemented by such means as would enable ecological economists in develop-

[4] It has been our policy to locate the workshops in isolated places, not in metropolitan centers, so that participants are not tempted away from the purpose at hand.

[5] The staff includes, in addition to Mäler and myself, Scott Barrett of the London Business School, John Dixon and Mohan Munasinghe of the World Bank, Carl Folke and Sara Anyar of the Beijer Institute, and Jeff Vincent of the Harvard Institute for International Development.

ing countries to enter the international network of *researchers*. If, as we economists tend to believe, there are "multiplier effects" in the dissemination of knowledge, the eventual size of the network could be expected to be a good deal larger. The Beijer Programme's second component consists of *research workshops*, at which participants from the South present their own research findings, works-in-progress, promising research proposals, and so forth, in the presence of foreign peers. My hope is that, through such workshops, our colleagues in developing countries will enjoy the benefit of peer-group evaluation and criticism of the kind that we, who are fortunate to work in advanced industrial countries, take for granted. These workshops are also held regionally, and their expenses are borne by the Beijer Institute.

Who attend research workshops? In order to build a comprehensive educational program, we have pursued the following approach: invitations to research workshops are sent to those who have previously attended a teaching workshop. This way, those who have obtained training at the teaching workshops have an incentive not only to teach the subject to students in their own universities, but also to conduct research on their local environmental problems. So we try to hold a research workshop some 12 to 18 months following each teaching workshop, and we invite previous participants to submit research papers for the occasion. To date, we have held seven research workshops: in Tanzania, Malta, Malaysia (twice), Venezuela, Costa Rica, and Zambia.

But even this is not enough. If capacity in ecological economics is to develop in the South, teaching and research workshops need to be supplemented by the creation of opportunities for indigenous scholars to publish their work in reputable international journals. The establishment, in 1996, of a quarterly journal, *Environment and Development Economics* (*EDE*), edited by Professor Charles Perrings of the University of York and published by Cambridge University Press, has been aimed at meeting this need. It forms the third component of the Beijer Programme.

The aim in starting the journal was to create a space where ecological and development economists from everywhere could publish their work. It is the intention of the Editorial Board to treat submissions from the South with the same care, sympathy, and critical standard that those scholars who work in the North take for granted. An important component of the journal is the Policy Forum section, where authors are being encouraged to report and comment on environmental debates and discussions currently being undertaken, most especially in their own countries. Economists in the South routinely learn about environmental problems of the North. It seems sensible that economists in the North are able to read about some of the many local environmental problems the poor in poor countries face regularly. Toward this, we will be using some of the grant money to help enable authors of the most promising papers to improve upon them so as to submit their work to *EDE*.

Intellectual infrastructure differs greatly within poor countries. Libraries in some universities subscribe to a wide range of international journals, whereas many others have an extreme paucity of subscriptions. Over an initial period of five years, the Beijer Institute is providing free subscriptions to *EDE* to libraries in the South

recommended to us by participants of our teaching and research workshops. In addition, the grant money for the Beijer Programme is being used to supply, free of charge, a selection of books on ecological economics to the libraries of the institutions where our participants belong. The books in question also form the basis of the teaching workshops.

The Programme's fourth component has only now begun: the creation of regional networks of ecological economists. There are already such networks, in Africa and Southeast Asia, to name two. We have initiated moves to establish a similar network among economists in South Asia.

Lessons

The Beijer Programme I have described here has several layers: it locates young university teachers of economics in poor countries (they are typically in their late 20s or early 30s) and tries to offer them training, encouragement, guidance, and incentives to both teach ecological economics to their students and to initiate research on their own. Toward this, we have tried several methods to locate suitable candidates for participation. We have had help from such organizations as the Consortium of African Economists; we have had help from our own academic contacts in the South; we have advertised the workshops and solicited applications; we have written to department heads in more than 150 universities, asking them to nominate young colleagues; and we have sought the help of past participants in locating able colleagues who are interested in ecological economics. Selection has on occasion proved to be a difficult exercise, but we have gained experience in this.

A typical teaching workshop of, say, 20 participants and 7 staff costs about US $120,000.00. Research workshops cost a bit less, because they are typically held over a 4–5-day period. Considering everything, this is likely to prove to be productive investment. Even if the success rate eventually were to turn out to be as low as 10% (e.g., even if only 10% of the participants in the teaching workshops were eventually to publish in *EDE*), it represents good value.

What have we learned from our experience? First, as expected, talent and background expertise vary across the continents, but there are excellent and concerned minds everywhere. Second, there is enthusiasm for the subject. Mäler and I had no idea that among our young colleagues in the South there is enormous interest in modern economics in general and in ecological economics in particular. In short, there is no need to "sell" ecological economics to our economics colleagues in the South: they want to teach and work on it anyway. Third, many of our colleagues in the South are not as yet dispirited about their prospects of teaching ecological economics and publishing articles on the subject. Fourth, our emphasis on teaching modern economic analysis to participants before getting down to details of ecological economics has, I believe, proved to be correct: participants have shown particular interest in seeing how even the most pedestrian problems concerning the environment are better informed when exposed to modern economic analysis.

Mäler's (and my own) prejudice was that it would be a mistake to restrict the lectures to the routine tools of ecological economics; we felt the right way to go about things would be to create a syllabus that encourages people to view ecological economics as an essential branch of economics. Judging by the response of most of our colleagues who have attended the teaching workshops, this prejudice has not been wrong. Fifth, the learning curve is steep. It has been a particularly gratifying experience to observe, again and again, the difference in the ability of the average participant to discuss environmental problems on the first days of a workshop and on the last day of the workshop.

The Future

What do these lessons tell us about possible future courses of action? Three recommendations suggest themselves: they involve the short, the medium, and the long run. Let me discuss them sequentially:

The immediate future

Of immediate concern is a request we have begun to receive from past participants. A number of them now want to prepare syllabi and reading lists for special courses on ecological economics they would like to establish at their own universities. They have asked the Beijer Institute for help in devising the appropriate syllabi. Since getting ecological economics taught in developing countries was the prime intention behind the Programme in the first place, we need to be able to provide this help. However, if we are to offer this help, we need funds to enable members of our staff to visit such universities as those that seek help, spend a few days understanding the existing economics program, address seminars in the presence of the senior colleagues there so as to legitimize our hosts' desires, and then advise our hosts on how best they could devise the syllabus. Fortunately, this involves small money. We could, for example, select a few key universities (totaling 6–7 in number in Asia, Africa, and Latin America) and help create teaching syllabi.

The medium term

To date, the Beijer programme has been supported by grants from the United Nations University's World Institute of Development Economics Research (UNU/WIDER), in 1992 and 1994; the Swedish International Development Authority (1995, 1998); and, more recently (1995–1998), from a block grant from the MacArthur Foundation. The latter grant will have enabled us not only to organize several teaching and research workshops, but also to finance some of the expenses associated with *EDE* and support the completion of research by some of the most promising submissions to *EDE* from developing countries. The MacArthur grant is being phased over four years. It started in 1995, peaked in 1997, and will tail off during 1998–1999. If the Beijer Programme is to have a lasting impact, it needs to consolidate

about 250 economists in the South into the network we are creating. But this means that it needs funding into the first two-to-three years of the new century.

The long term

The emergence of the Internet will prove to be very important in enabling our colleagues in the South to join the international academic network. But in the final analysis, I believe nothing can replace personal contact. You have to be in close association with colleagues in the South to realize how separated many feel from the kind of collegiate contact, encouragement, and help that makes academic life creative in the West. Many years ago, the late Abdus Salam recognized the importance of this and created, with the help of UNESCO and the Italian Government, among other agencies, the International Centre for Theoretical Physics in Trieste. A number of physicist friends of mine working in the South have told me how important their visits to this Institute have been in enabling them to "re-charge" themselves intellectually, during sabbaticals and vacations. Over the long run, I am convinced we need to establish something like such an institute, but located in a university campus in the South.

What I have in mind is an institute that would have a skeletal staff and serve visitors. The Institute's purpose would be to facilitate colleagues in the South in their attempts to teach and conduct research at the interface of anthropology, ecology, economics, demography, geography, and political science, which is where the subject of ecological economics resides.[6] The way the Institute could facilitate scholars would be by offering financial aid for visits to it during vacations, sabbaticals, and so forth. The Institute would also invite internationally distinguished colleagues in these same fields to come on short-term visits to deliver lectures, conduct seminars, and generally interact with the Institute's prime visitors. In other words, the Institute's mission would be to offer an intellectual center for scholars from the South. Teaching and research workshops, such as those in the Beijer Programme, could also be located at the Institute.

Vast sums are spent annually on higher education in poor countries. Minor rechanneling of resources could greatly improve the performance of those who teach in these places. Capacity building involves a wide range of activities. The Beijer Programme that I have described in this article is a very small part of this, but all my experience tells me now that it is an important part. Fortunately, it is relatively speaking, an inexpensive part.

Acknowledgments

This note was presented at the conference on "Managing Human-Dominated Ecosystems," held at the Missouri Botanical Garden, St. Louis, during 26–28 March 1998. It is based on a paper presented at the Swedish International Development Authority's Seminar on Research Cooperation Towards 2015, during 4–5 November 1996 in Stockholm. I am greatly indebted to Karl-Göran Mäler for collaboration over many years on matters discussed here.

Literature Cited

Agarwal, A. & S. Narain. 1989. Towards Green Villages: A Strategy for Environmentally Sound and Participatory Rural Development. Centre for Science and Environment, New Delhi.

Anderson, D. 1987. The Economics of Afforestation.Johns Hopkins Univ. Press, Baltimore.

Barrett, S. 1989. Economic Growth and Environmental Conservation. Ph.D. Dissertation, London School of Economics.

Binswanger, H. 1991. Brazilian policies that encourage deforestation in the Amazon. World Development 19(7): 821–829.

Bojö, J., K.-G. Mäler & L. Unemo. 1990. Environment and Development: An Economic Approach. Klewer, Amsterdam.

Cropper, M. & W. Oates. 1992. Environmental economics: A survey. Journal of Economic Literature 30(2): 675–740.

C.S.E. 1982, 1985. The State of India's Environment: A Citizen's Report. Centre for Science and Environment, New Delhi.

Dasgupta, P. 1982. The Control of Resources. Harvard Univ. Press, Cambridge, Massachusetts.

Dasgupta, P. 1993. An Inquiry into Well-Being and Destitution. Clarendon Press, Oxford.

Dasgupta, P. 1995. The population problem: Theory and evidence. Journal of Economic Literature 33(4): 1979–1902.

Dasgupta, P. 1996. The economics of the environment. Environment and Development Economics 1(4): 387–428.

Dasgupta, P. & K.-G. Mäler. 1991. The environment and emerging development issues. Proceedings of the Annual World Bank Conference on Development Economics 1990 (Supplement to the World Bank Economic Review and the World Bank Research Observer), 101–131.

Dasgupta, P. & K.-G. Mäler. 1995. Poverty, institutions and the environmental resource-base. *In:* J. Behrman & T. N. Srinivasan (editors), Handbook of Development Economics, Vol. IIIA. North Holland, Amsterdam.

Dasgupta, P. & K.-G. Mäler (Editors). 1997. The Environment and Emerging Development Issues, Vols. 1 & 2. Clarendon Press, Oxford.

Dixon, J. A., D. E. James & P. B. Sherman. 1989. The Economics of Dryland Management. Earthscan Publications, London.

Dixon, J. A., D. E. James & P. B. Sherman (Editors). 1990. Dryland Management: Economic Case Studies. Earthscan Publications, London.

Dixon, J. A. et al. 1994. Economic Analysis of Environmental Impacts. Earthscan Publications, London.

Ehrlich, P. R., A. H. Ehrlich & G. C. Daily. 1995. The Stork and the Plow: The Equity Answer to the Human Dilemma. G.P. Putnam's Sons, New York.

Falconer, J. & J. E. M. Arnold. 1989. Household Food Security and Forestry: An Analysis of Socio-economic Issues. FAO, Rome.

Solorzano, R. et al. 1991. Accounts Overdue: Natural Resource Depreciation in Costa Rica. World Resources Institute, Washington, D.C.

World Bank. 1992. World Development Report 1992. Oxford Univ. Press, New York.

World Commission. 1987. Our Common Future. Oxford Univ. Press, New York.

Industrial Ecology

Iddo K. Wernick
Columbia Earth Institute,
405 Low Library,
New York, New York 10027, U.S.A.
iw4@columbia.edu

Abstract

Similar to the study of natural ecosystems, industrial ecology studies human systems of industrial production and consumption by analyzing the flow of material and energy through them. The purpose of this type of analysis is to better characterize industrial systems in order to identify technological, policy, and economic instruments that reduce the resource use and waste generation of industrial society. Several examples are considered showing material flows at the level of a single industrial complex and for the United States economy as a whole. Further examples look at the resource flows and toxic emissions associated with the consumption of lead and forest products in the United States.

I. Introduction

Industrial ecology studies all material flows among different industries, consumers, and the environment. Like biological ecologists, industrial ecologists analyze systems and their mechanisms. Industrial ecology asks two essential questions: How can industry and society develop food webs that re-circulate materials within the industrial system to resemble those found in natural ecosystems? And, How can industry and society lessen their environmental impact while providing for human needs and wants?

To answer these questions, industrial ecology analyzes the flows of various materials in the economy, distinguishing industrial *systems* from their *artifacts*, and the design of more environmentally benign products. Using materials and energy flows as organizing principles, industrial ecology searches industrial systems for

technical and management options that promise less use of virgin natural resources and less pollution released to the environment. Like biological ecologists, industrial ecologists strive to understand systems using the insight provided by examining the material and energy streams that flow through them.[1]

Both large- and small-scale material flows impact the environment. Anthropogenic flows of hydrocarbon fuels for energy affect global biogeochemical cycles that govern climate and ecosystems. Forestry and agricultural management affect land use, the potential for carbon uptake in terrestrial ecosystems, and nutrient and sedimentary flows at the regional to continental level. Toxic emissions, though far smaller in volume, can flow into municipal drinking water supplies, enter the human food chain, and upset the biological balance of natural ecosystems.

In the larger social and historical context, the shift toward more knowledge-based economies in varying degrees around the globe does not necessarily portend a civilization where consumption of industrial materials is obsolete. People using computers still require shelter, transportation, food, energy, and other basic amenities that rely on the mobilization of vast amounts of materials. Rather, greater knowledge offers opportunities for humans to better understand and better manage the flow of materials from nature to industry and society as well as the flow of industrial and consumer residuals to nature. Specifically, greater knowledge allows for better characterization of natural and industrial systems to yield better technical solutions and better economic and policy tools to reduce the resource use and pollution that affect ecosystems

Although humanity's effects have necessarily become important to the study of non-human ecosystems, ecologists rarely examine the details of the human technological systems that lay behind them. Industrial ecology supplies the analysis of how materials cycle within the human system and opportunities to improve the industrial resource use and reduce the wastes that damage ecosystems.

II. Industrial Ecosystems

Looking at the material relations within and between industries forms one aspect of industrial ecology study. The historical progression of material and energy usage in the petrochemical industry provides a prime example of increasing material and energy efficiency. By capturing the residual heat and distillates from unit operations, refineries have steadily increased the value they derive from a barrel of oil. Other industries such as the paper and primary metals industry have also developed processes to maximize value recovery from material and energy resources. While the volume of materials processed in these industries has grown steadily over this century in the United States, the amount of pollution per unit of production, and in some cases absolute amounts, have fallen.[2] Knowledge-driven changes in industrial practice can thus engender greater resource efficiency and also help reduce environmental burden.

Greater resource efficiency crosses industrial sectors as well. The Industrial Ecopark at Kalundborg, Denmark, offers a prototype of material and energy ex-

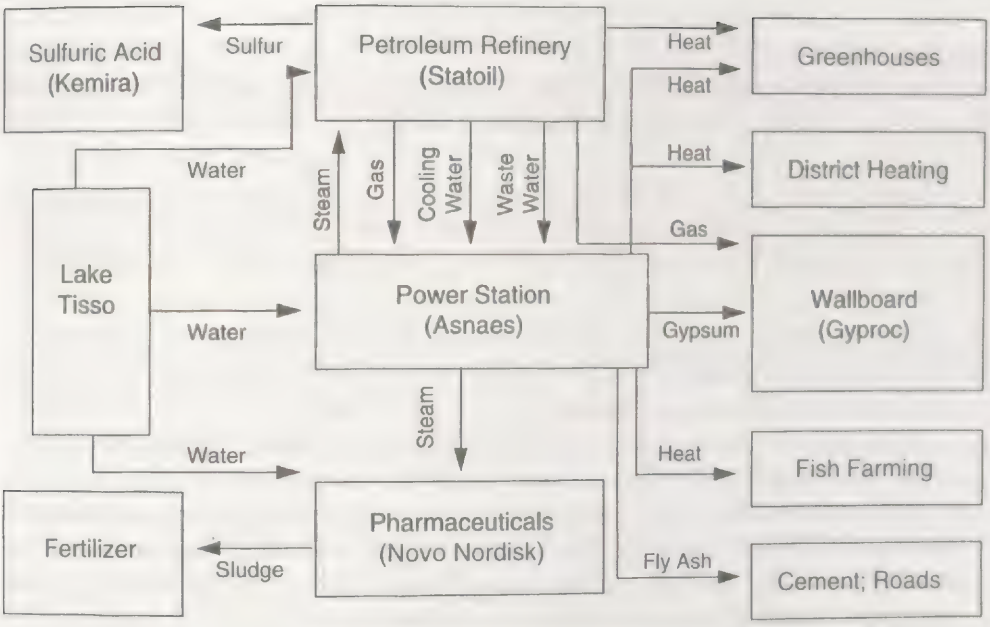


Figure 1. The Industrial Ecopark at Kalundborg, Denmark.

change between different industries co-located in the same industrial complex. The park houses a petroleum refinery, power plant, pharmaceutical plant, wallboard manufacturer, and fish farm that have established dedicated streams of processing wastes (including heat) between individual facilities. Figure 1 shows a schematic diagram of the Kalundborg Industrial Ecopark.[3]

Examples of industrial symbiosis are not limited to fixed central facilities like Kalundborg. Due to sunken capital and a host of other factors, widespread establishment of Ecoparks seems more of a dream than a pragmatic strategy for the near term. Recovering the resource value of process wastes from industry will require the existence of markets for waste materials that offer firms the opportunity to buy and sell wastes over distances. International trade in scrap metals, which exceeded 40 10^6 tons in 1995, provides an example.[4] Elements of this market for wastes that need to be replicated for other waste materials include the establishment of effective pricing mechanisms, transportation networks, and technologies that accommodate secondary feedstocks in production. Inescapably, the price difference per ton of primary and secondary feedstock plays a decisive role in determining the economic viability of the secondary market. Governments and industries in both the United States and Europe have conducted experiments with waste exchanges with mixed results.[5] These experiments and other studies[6] show that government policies that favor production from virgin materials or that penalize trade and transport of secondary materials tilt the "playing field" away from secondary materials industries. Thus, even without direct government intervention to incentivize material recycling, policies that remove economic disincentives to secondary materials markets will improve their economic viability.

III. National Material Flows

From a more macroscopic perspective, industrial ecology seeks to characterize material flow at the national level. Material flows for a nation indicate the configuration and extent of national industrial activity, representing environmentally important national enterprises such as energy production, mining, agriculture, forestry, and construction. Human wants also map to national materials consumption as people consume materials to provide shelter, food, power, mobility, and contact with others.

Figure 2 shows a schematic view of the size and structure of the material flows in the United States in 1990.[7] The framework considers primarily three components: inputs to the economy (including imports), outputs (including exports), and extractive wastes. Extractive wastes are grouped separately because they represent immense mobilizations of materials readily distinguished from commodities, products, and other wastes. Inputs are divided into energy, construction minerals, industrial minerals, metals, forestry products, and agricultural products and outputs as domestic stock (the built environment), atmospheric emissions, dissipation, other wastes, and recycled materials. Imports and exports represent the masses of major individual commodities and classes of commodities crossing U.S. borders. The values for extractive wastes, which include residues from the mining and oil and gas industries, and water consumption are accounted for in the figure to show the sheer weight of these waste flows located at the base of the industrial food chain and their relation to others. The figure omits consumption of atmospheric oxygen for biological respiration and in industrial processes but does include atmospheric nitrogen fixed for fertilizers and oxidized in industrial processes. Finally, the figure does not explicitly consider manufactured chemical products, but does include the mass of feedstocks used for organic and inorganic chemical production.

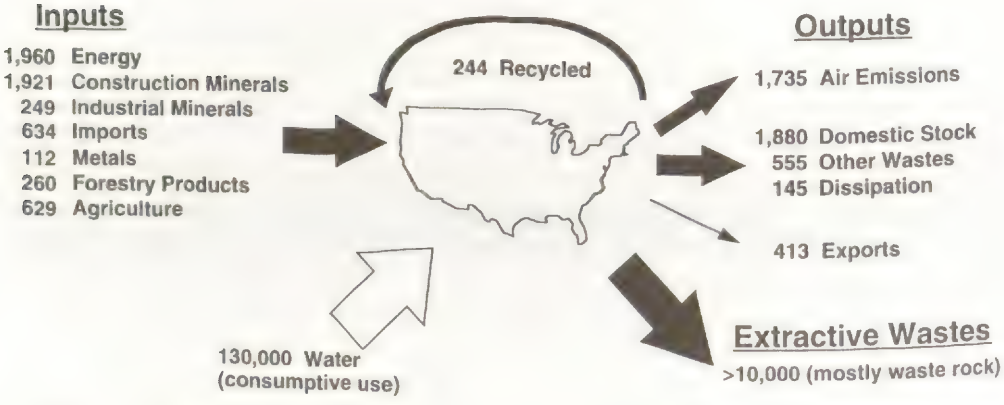


Figure 2. U.S. material flows, ca. 1990. All values in 10^6 metric tons. Consumptive water use is defined as water that has been evaporated, transpired, or incorporated into products, plant, or animal tissue, and is therefore unavailable for immediate reuse.

One of the drawbacks of aggregated national material accounts is the use of a single unit of measure, mass, which is necessary to present the data in a uniform manner and allows for comparing the relative size of different material flows. However, kilograms and tons can hide the variables that impact ecosystems most such as toxicity. For instance, the threat posed by unregulated emissions of persistent organic compounds like dioxin and furan are significant, while the mass involved is on the order of kilograms to tons.[8] Nevertheless, weight provides a reasonable starting point for appreciating the structure and scale of major activities affecting national environmental quality. Furthermore, national mass flow accounts allow for cross-national comparisons and provide the basis for rudimentary measurement of "national ecosystem performance." Table 1 shows sample metrics for evaluating

Table 1. National Material Flows: Sample Metrics.

Metric	*Dimensions*	*Formula*	*Environmental significance*
Total Inputs	MMT and MMT/Capita	Aggregate consumption of material by class on absolute & per capita basis	Benchmarking national resource use
Fuel mix (H/C)	Dimensionless	Consumption ratio of Btu's from Gas:Petroleum:Coal	CO_2 emissions
Intensity of Use	MMT of inputs/$GDP	Material consumption quantity for selected input materials/GDP	Relationship of resource use to economic activity
Agricultural intensity	Dimensionless	Mineral consumption for Agriculture/Total crop production	Eutrophication, soil erosion, ecosystem disruption, energy use
Decarbonization	MMT of carbon inputs/$GDP	Carbon inputs/GDP	Relationship of greenhouse emissions to economic activity
Raw materials index	%	Raw material extraction/ Total material inputs	Materials efficiency, recycling, mining and processing waste, energy use
Metals recycling rate	%	Secondary metals production/Primary production from ores	Materials efficiency and recycling
Renewable net carbon balance	%	Forest growth/Forest products harvested	Global carbon balance, land use, ecosystem disruption
Green productivity	%	Solid wastes/Total solid physical outputs	Materials efficiency and recycling
Intensity of use for residues	MMT/$GDP	Selected materials waste streams/GDP	Relationship of waste generation to economic activity
Leak index	MMT	Quantity of materials dissipated into the environment	Materials efficiency, recycling, media contamination
Process to post-consumer wastes ratio	%	Process wastes from industry/Post-consumer waste	Relating hidden to apparent consequences of industrial production
Physical Trade Index	%	Mass value of net trade in manufactured products/ Mass value of net trade in raw resources	Domestic and foreign resource consumption & pollution
Mining byproduct recovery	Dimensionless	Total byproduct recovery/ Total output	Materials efficiency, solid wastes

* MMT=10^6 Metric Tons, GDP = Gross Domestic Product.

national resource use, waste generation, industrial resource efficiency, and significant leaks of material to the environment.[9]

IV. Specific Flows

In this section we look at the material flow of lead and forest products in the U.S. economy. The manner in which these materials are extracted, processed, and consumed affect local, and in some cases global, ecosystem health. Lead is a toxic metal demonstrated to threaten human health as well as the health of ecosystems through accumulation in soils and living organisms. Timber, on the other hand, represents an industrial material that has no toxic characteristics per se, but whose harvest from forests can threaten wildlife populations and the ability of forest ecosystems to store carbon and retain water. Also, both lead and wood are fundamental to industrial societies as they are essential to human needs including housing (lumber), transport (batteries), and communication (paper).

Lead

Annual consumption of lead in the United States is approximately 1.5 10^6 metric tons, approximately 60% of which is secondary production.[10] Like other toxic metals, the physical form of lead emissions as well as the media into which they are emitted are central to evaluating their threat to human and ecosystem health. In the United States, the single most important environmental change in the use of lead came with the removal of lead from gasoline in the early 1980s. By the early 1970s, the use of lead as an additive to gasoline reached a global peak of over 400 10^3 metric tons, falling to under 70 by 1993.[11] The removal of lead from gasoline not only reduced annual production and consumption of lead, but removed a dispersive source of this toxic element.

Recent debate about lead has focused on the bioavailability of lead compounds from the emissions of secondary lead smelters, which form the heart of the automotive lead acid battery recycling system. In 1995, Lave et al.[12] argued that the expanded secondary production of lead associated with their use in electric cars would result in higher lead emissions per kilometer than peak lead emissions from gasoline. Socolow and Thomas[13] countered that the most hazardous form of release, air emissions, from automobile battery lead smelters summed to only 57 metric tons in 1994 in the U.S. The corresponding "emission factor" (tons emitted per ton produced), 6 10^{-5} for the U.S. in 1994, exceeds those available with new processes that go as low as 1 10^{-5}.[14] The other loss of lead in the smelting process is as lead oxide dissolved in the silicate slag produced during the smelting of batteries for which recovery processes are available.[15]

Figure 3 shows both the considerable progress in closing the cycle of lead in the U.S. as well as the leaks in the system (in the lower right-hand corner) where lead flows remain largely open-ended, and invite strategies to close them.

Note that although the land used for extracting primary metals represents under 0.1% of Earth's terrestrial surface,[16] the necessary infrastructure for mining activity and the waste flows to soil, water, and air affects surrounding ecosystems.

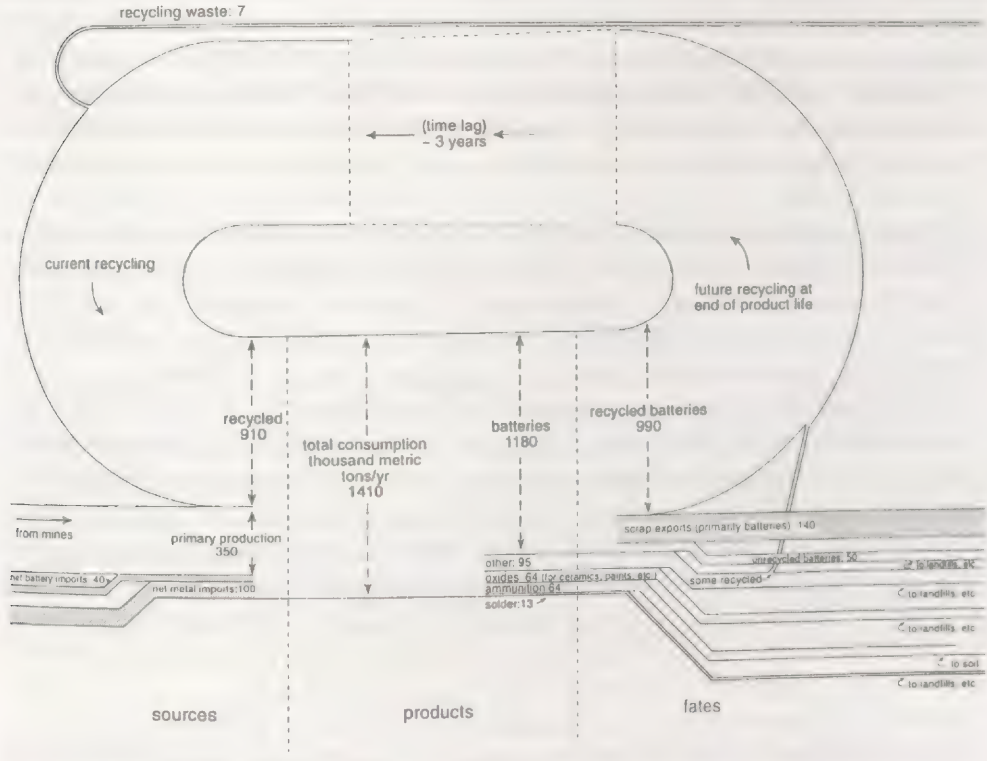


Figure 3. Flows of lead through the U.S. economy, 1993–1994. (Data displayed are averages of data for 1993 and 1994, in thousands of metric tons of lead per year.) "Other products" are bearings, brass, bronze, cable covering, caulking lead, casting materials, pipes, and sheet lead. Emissions to water or air—the primary health concerns—are not shown because they are too small to be visible at this scale. Also not shown is the roughly 0.6 thousand metric tons of lead used in aviation gasoline (about a billion liters with a lead concentration of 0.4–0.8 grams per liter). Courtesy of Socolow and Thomas (1997).

Forest products

Forest products are unique in that their provision directly ties human needs to ecological disruption through the use of land. In the U.S., the timber removed annually from the forest weighs more than twice the annual consumption of all metals combined.[17] In less industrialized countries, the use of wood for fuel and land clearing for shifting agriculture mean that wood occupies an even greater share of the national materials diet.

Though earlier in their history the U.S. and other developed countries cleared vast amounts of forest land for agriculture, human settlements, and transportation

corridors, today the industrialized countries of the world primarily use wood directly for products. Forests in the OECD countries have generally grown in extent in the last decades.[18] From 1952 to 1993 the carbon stored in U.S. forests grew one third, adding close to 9 10^9 metric tons, equal to more than six years' present U.S. emissions of carbon from burning fossil fuels.[19] In many parts of the world, however, the extent and volume of forests continue to fall. To persuade more environmentally sound land use in poorer countries, the effort to place value on forest ecosystem services should offer tangible economic opportunities and conditions that allow people to subsist from a given area of land without relying on the need to remove timber.

The following analysis of the material flow of wood in the United States does not directly address the causes of forest ecosystem disruption in other parts of the world. However, it presents a model where a climate of economic prosperity and societal concern have pushed industry to become more efficient in obtaining maximal value from resources and reducing emissions to the environment.

Each phase of activity from consumption and disposal back to processing and forestry contributes to the human impact on the forest. Shifts in construction and manufacturing, changes in consumer practice, and regulations that determine timber availability all influence what will be harvested and where.[20] Foresters help determine the short-term impact of logging and the long-term adequacy of timber supplies. The ability of saw and paper mills to convert incoming timber into products more efficiently allows them to diminish the need to fell more trees. Finally, the decision whether to burn, bury, or buy discarded forest products as recyclables modifies the amount of timber that must come from the forest and determines whether these material flows burden society or constitute a resource.

Pollution from the forest products industry fell over the last decades. To reduce emissions, paper mills have responded to government environmental mandates with technologies such as electrostatic precipitators, scrubbers, and advanced water treatment systems.[21] U.S. paper mills have reduced Biological Oxygen Demand per ton of paper produced by 80–90% and releases of total suspended solids by about three quarters since 1959. Much of the pollution reduction comes from more efficient water use with newer mills recycling process water up to seven times. Pressure to lower emissions of some chlorinated compounds has spurred research into substitutes for elemental chlorine for bleaching paper including peroxide processes, the use of chlorine dioxide (ClO_2) to reduce dioxin formation, and alternatives to sodium chlorite (NaClO) to reduce chloroform generation. From 1988 to 1996, toxic releases, as reported in the U.S. EPA's annual Toxics Release Inventory for the Paper Sector (Standard Industrial Classification (SIC) 26) fell 17%.[22] For wood products (SIC 24), releases fell 18% over the same period, with wood preservatives, such as creosote and toluene, used to extend product life in the economy, forming the largest volume of toxic wastes.

Figure 4[23] presents a summary of the material flows in the U.S. wood products industry by volume. The left side of the figure shows that 646 10^6 m^3 of wood

entered the U.S. economy in 1993. Trees furnished 78%, recycling 10%, and imports 12%. The categories of Other Removals (timber removed to thin stands or clear land) and Logging Residues (wood left on the forest floor) never enter the commercial flow, although their relative quantity is shown by the size of the rectangles. The right side of the figure shows that 26% was consumed as solid wood, 26% as paper, 36% as fuel, and 10% exported. The center column of the figure shows wood processing. For example, saw mills received 200 10^6 m^3 of timber to produce 84 10^6 m^3 of lumber. The dotted lines show the residue flows from sawn lumber and composite materials into other mill processes, especially paper-making and fueling the mills themselves.

The value of saw mill products declines from solid lumber, to plywood and composite products made from glued chips, strands, and particles, and finally to chips

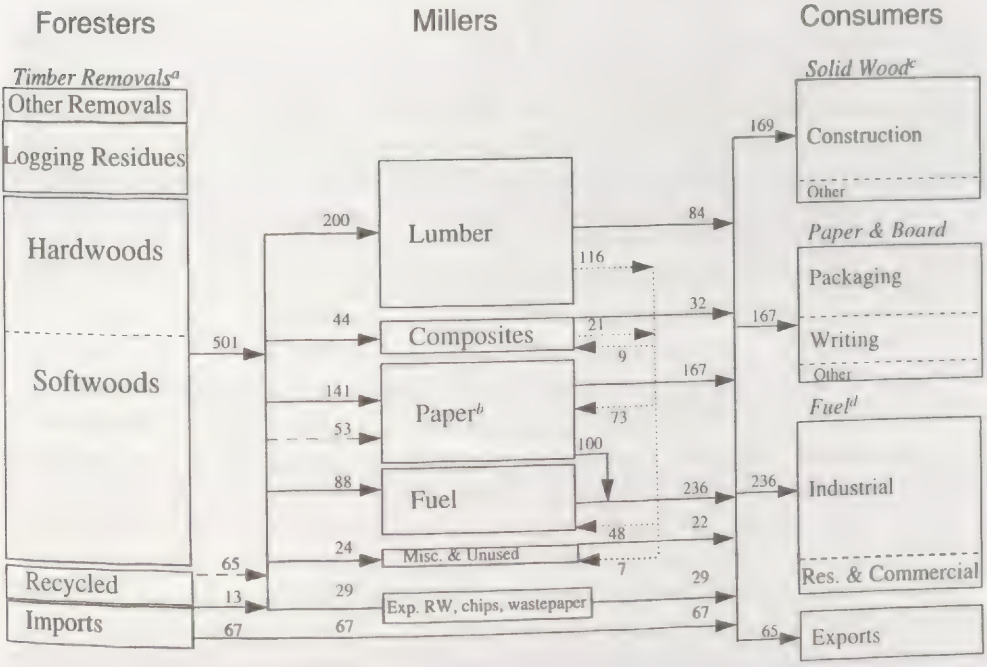


Figure 4. Material flows in the U.S. forest products industry, 1993. Box heights are to scale. All values in 10^6 m^3. For paper we consider 1 metric ton to be equivalent to 2 m^3. (a) Based on the ratio of logging residues (15.1%) and "Other Removals" (6.6%) to all removals for 1991. (b) The dashed line entering paper represents the inputs from "recycled." We estimate that 100 10^6 m^3 of the woody mass entering paper mills undergoes combustion for energy. (c) Construction includes millwork such as cabinetry and moldings. "Other" includes industrial uses such as materials handling, furniture, and transport. (d) The ratio of end uses relies on Btu data from the U.S. DOE Energy Information Administration. The category "Residential and Commercial" includes Electric Utilities. Sources: Ince (1994); Energy Information Administration (1994); U.S. Department of Agriculture (1993); U.S. Bureau of the Census (1995); Amer. Forest & Paper Assoc. (1995); Smith et al. (1994[12]); and data from the Engineered Wood Products Assoc., Tacoma, Washington, and the Western Wood Products Assoc., Portland, Oregon.

and dust for paper or fuel. Prime strategies in the industry are to get more of the valuable lumber from logs and to find uses for irregular shapes, waste, and unused species of trees. Industrial research and collaborations with government bodies such as the U.S. Forest Product Laboratory have yielded useful innovations. Best Open Face technology optimizes the initial line for sawing[24] and sharper, more stable blades can increase lumber recovery from logs by 5–10%.[25] The various improvements have caused the ratio of lumber to roundwood to rise from 33% in 1970 to 42% in 1993.[26,27] The need to obtain rectangular boards from round logs finally limits the yield of sawn lumber, at least until geneticists engineer square trees.

Better blades allow the plywood industry to peel smaller diameter logs from younger trees and to peel all logs to smaller diameters to get more product per log.[28] Using wood residues and organic adhesives, mills produce more varied and higher quality composite products that compete with solid lumber in construction and industry. Composites like parallel laminated veneer, medium density fiberboard, and insulation board form points on a continuum of products from solid lumber to paper that use wood and fiber.[29] Particleboard, made from residues, and Oriented Strand Board, made from strands of previously unused aspen and other species, were largely absent from the inventory of wood products in 1970. By 1993, the composites plus plywood comprised 28% of solid wood products, up from the 19% comprised of plywood alone in 1970. Several of these products can also use some of the tens of millions of cubic meters of discarded wood and waste paper in U.S. municipal solid waste.[30,31]

Paper mills use pulpwood harvested from the forest, wood residues, and waste paper to make their product. Since 1970, residue input to paper manufacture rose over 70% and recovery of wastepaper for recycling rose 150%.[32,33] Thus, despite a 64% increase in U.S. paper and paperboard production between 1970 to 1993, the amount of wood from forests entering paper mills rose only about 16%. Using wood bark and process liquors for fuel, U.S. paper mills generated more than half their energy from process byproducts by the early 1990s.[34] Finally, taking advantage of more abundant species, the use of hardwoods to make pulp rose from about a quarter of all pulp used for paper in 1976 to a third in 1990.[35,36]

Because wood used for fuel has no dimensional requirements, modern mills can approach 100% materials efficiency by collecting and burning residues. Using fuelwood from the forest and wood residues from manufacturing, U.S. lumber and plywood mills generated over 72% of their energy internally by 1991.[37] The percentage of timber harvested for fuel has remained relatively stable since the beginning of the 1980s after rising fourfold since 1970 when the Oil Shocks induced it to rise to about 15% of the total harvest.[38,39] Currently, about three-quarters of the wood harvested for fuel comes from trees considered unsuitable for other products.[40] Though the removal of any trees can adversely affect the ecosystem services that forests provide, judiciously harvesting trees for fuelwood can have salutary effects on the forests by yielding space, sunlight, and nutrients to more desired species. Including the volume of fuelwood, burned residues, and pulping liquors, 47% of the U.S. timber harvest in 1993 became useful energy.

Compared to 1970 processors of forest products collectively spared about 120 10^6 m^3 of roundwood harvest annually that would have otherwise been used without improved efficiency. Over the decade of the 1980s, annual timber harvest in the U.S. affected about 4 10^6 hectares on average (estimate by W. Brad Smith, U.S.D.A. Forest Service, 1997). Because roundwood removals were about 500 10^6 m^3 over the decade, loggers obtained an average of roughly 125 m^3 of timber for roundwood products for every hectare harvested. As a result, one could say that greater process efficiency allowed the forest products industry to leave 960,000 acres of land undisturbed.

Though forest conservation will require sustained diligence from both the scientific and government communities, U.S. forests already provide an example of changing industrial and social practice to reduce pressure on ecosystems.

V. Conclusions

Much remains to be developed in applying and testing the analogy between natural and industrial ecosystems. Industrial ecology can learn from biological ecology about metrics of system performance for systems containing a diverse set of actors and a multiplicity of material and energy flows. Such metrics allow the field to move beyond describing and cataloging systems to developing and testing hypotheses. Simple system descriptions can nonetheless reveal opportunities in some cases. For example, a material accounting of silver wastes performed at UCLA identified photography labs and dentist's offices as the major source of high silver levels in the San Francisco Bay.[41] Finally, numerical accounts of material flows offer the general benefit of providing governments and corporations with data that can be more readily integrated into traditional policymaking and accounting frameworks.

The goal of industrial ecology is understanding what economic, technological, and organizational tools will be necessary for the establishment of an industrial system that serves the needs of humankind while operating within the context of a healthy global ecosystem. Most current industrial systems require a high throughput of energy and materials to function and exhibit little or no resource recovery. Some systems have reached the transitional stage where resource recovery becomes more integral to the workings of the system but still fall short of satisfying input requirements for resources. Ultimately, industrial ecology envisions a system that relies exclusively on external energy inputs while cycling all of the material outputs of production. Achieving this objective begins with analyzing present technological systems and searching for alternatives that limit resource requirements and recover waste products. In this search, industrial ecology looks to natural ecosystems for inspiration, seeking both to emulate them within industry and to protect them from disruption.

Acknowledgments

The author wishes to thank Jesse Ausubel of The Rockefeller University and Paul Waggoner of the Connecticut Agricultural Experiment Station with whom he collaborated on the forest analysis and who helped develop the ideas presented here.

Literature Cited

[1] Graedel, T. E. 1996. On the concept of industrial ecology. Annual Review of Energy and Environment 21: 69–98.

[2] INFORM. 1995. Toxics Watch 1995. INFORM, New York.

[3] Graedel, T. E. & B. R. Allenby. 1995. P. 287 *in* Industrial Ecology. Prentice Hall, Englewood Cliffs, New Jersey.

[4] Sibley, S. F., W. C. Butterman & Staff. 1995. Metals recycling in the United States. Resources, Conservation, and Recycling 15: 259–267.

[5] U.S. Environmental Protection Agency. 1994. Review of Industrial Waste Exchanges. Report # EPA-530-K-94-003, Waste Minimization Branch, Office of Solid Waste, USEPA, Washington, D.C.

[6] Federal Disincentives: Study of Federal Tax Subsidies and Other Programs Effecting Virgin Industries and Recycling. 1994. EPA Report no. EPA230R94005, National Service Center for Environmental Publications, Cincinnati, Ohio.

[7] Wernick, I. K. & J. H. Ausubel. 1995. National materials flows and the environment. Annual Review of Energy and Environment 20: 462–492.

[8] Thomas, V. M. & T. J. Spiro. 1995. An estimation of dioxin emissions in the United States. Toxicological and Environmental Chemistry 50: 1–37.

[9] Wernick, I. K. & J. H. Ausubel. 1995. National material metrics for industrial ecology. Resources Policy 21(3): 189–198.

[10] U.S. Geological Survey. 1997. Mineral Commodity Summaries. U.S. Government Printing Office, Washington, DC.

[11] Socolow, R. & V. Thomas. 1997. The industrial ecology of lead and electric vehicles. Journal of Industrial Ecology 1(1): 13–36.

[12] Lave, L. B., C. T. Hendrickson & F. C. McMichael. 1995. Environmental implications of electric cars. Science 268: 993–995.

[13] Ibid.

[14] Karlsson, S. 1996. Can Metals be Used Sustainably? The Example of Lead. Annual Report, Institute of Physical Resource Theory, Chalmers University of Technology and Goteborg University, AFR report no. 121. Swedish Environmental Protection Agency, Stockholm, Sweden.

[15] Brew, R. B. M., C. R. Fountain & J. Pritchard. 1991. ISASMELT for secondary lead smelting. Pp. 170–181 *in* Lead '90, London. Lead Development Association.

[16] Barney, G. O. 1980. The Global 2000 Report to the President of the United States. Pergamon, New York.

[17] See reference 6.

[18] World Resources Institute. 1995. World Resources 1994–5. Oxford Univ. Press, New York.

[19] Birdsey, R. A., A. J. Plantinga & L. S. Heath. 1993. Past and prospective carbon storage in United States forests. Forest Ecology and Management 58: 33–40.

[20] Ince, P. J. 1995. What Won't Get Harvested Where and When: The Effects of Increased Paper Recycling on Timber Harvest. Working Paper #3. School of Forestry and Environmental Studies, Yale University.

[21] Armstrong, A. D., K. M. Bentley, S. F. Galeano, R. J. Olszewski, G. A. Smith & J. R. Smith. 1998. The pulp and paper industry. Pp. 101–141 *in* D. J. Richards & G. Pearson (editors), The Ecology of Industry: Sectors and Linkages. National Academy Press, Washington, D.C.

[22] U.S. Environmental Protection Agency. 1998. 1996 Toxics Release Inventory Public Data Release. EPA publication 745-R-98-005, USEPA, Washington, D.C.

[23] Wernick, I. K., P. E. Waggoner & J. H. Ausubel. 1998. Searching for leverage to conserve forests: The industrial ecology of wood products in the U.S. Journal of Industrial Ecology 1(3): 125–145.

[24] U.S. Congress Office of Technology Assessment. 1984. Wood Use—U.S. Competitiveness and Technology Volume II: Technical Report. Office of Technology Assessment Report OTA-M-224. Washington, D.C.

[25] U.S.D.A. Forest Service. 1982. Analyses of the Timber Situation in the U.S. 1952–2030. U.S. D.A. Forest Service Forest Resource Report 23. Pp. 264–272 and table 3.68.

[26] Grantham, J. B. & J. O. Howard. 1980. Logging residue as an energy source. *In* K. V. Sarken & D. A. Tillman (editors), Progress in Biomass Conversion. Academic Press, New York.

[27] Ince, P. J. 1994. Recycling of Wood and Paper Products in the United States. A Report for the UN Economic Commission for Europe Timber Committee Team of Specialists on New Products, Recycling, Markets and Applications for Forest Products. United States Department of Agriculture, Forest Products Laboratory, Madison, Wisconsin.

[28] See reference 24.

[29] Maloney, T. M. 1996. The family of wood composite materials. Forest Products Journal 46(2): 19–26.

[30] English, B. 1994. Wastes into wood: Composites are a promising new resource. Environmental Health Perspectives 102(2): 168–170.

[31] Skog, K. E., P. J. Ince, D J. S. Dietzman & C. D. Ingram. 1995. Wood products technology trends. Journal of Forestry 93(12): 30–33.

[32] American Forest & Paper Association. 1995. 1995 Statistics: Paper, Paperboard, & Woodpulp. American Forest & Paper Association, Washington, DC.

[33] U.S. Bureau of the Census. 1995. Statistical Abstract of the United States. 115th ed. U.S. Government Printing Office, Washington, D.C.

[34] Ibid.

[35] U.S.D.A. Forest Service. 1982. Analyses of the Timber Situation in the U.S. 1952–2030. U.S.D.A. Forest Service Forest Resource Report 23. Pp. 264–272 and table 3.68.

[36] Haynes, R. W., D. M. Adams & J. R. Mills. 1995. The 1993 RPA Timber Assessment Update. U.S.D.A. Forest Service General Technical Report RM-GTR-259.

[37] Energy Information Administration. 1994. Manufacturing Consumption of Energy. U.S. Department of Energy Report DOE/EIA-0512(91). U.S. Government Printing Office, Washington, D.C.

[38] Powell, D. S., J. L. Faulkner, D. R. Darr, Z. Zhu & D. W. MacCleery. 1993. Forest Resources of the United States, 1992. U.S.D.A. Forest Service General Technical Report RM-234.

[39] See reference 25.

[40] U.S.D.A. 1993. Agricultural Statistics 1993. U.S. Government Printing Office, Washington, D.C.

[41] Kimbrough, D., P. Wong, J. Biscoe & J. Kim. 1996. A critical review of photographic and radiographic silver recycling. Journal of Solid Waste Technology and Management 23(4): 197–207.

[42] Smith, W. B., J. L. Faulkner & D. S. Powell. 1994. Forest Statistics of the United States, 1992. USDA Forest Service Report GTR-NC-168.

Markets and Sustainability

Geoffrey Heal[1]
Columbia Business School
616 Uris Hall, 3022 Broadway
New York, New York 10027-6902, U.S.A.

Abstract

Do markets lead us to make sustainable choices? If not, why not? And what would we need to do to remedy this? This paper takes a preliminary look at these questions. It identifies three categories of reasons why market choices may not be sustainable, related to valuation of the future, recognition of the benefits provided by environmental assets, and incorrect incentives. It gives examples of cases in which these problems have been corrected, and considers the scope for a more positive relation between market forces and conservation of the environment.

Introduction

A good way to start talking about sustainability is by providing a definition. There is no generally agreed-upon definition, so let me suggest one that contains the essence of much of the discussion in the literature. Essentially, sustainability is choosing to do things that we can safely continue indefinitely: choosing actions and ways of operating that can be continued over long periods without unacceptable consequences, or without unacceptable risks of unacceptable consequences. This idea seems to capture much of what is implicit in general discussion.[2] Most

[1] I am grateful to G. C. Daily and G. D. Tilman for many conversations about the issues discussed here: the influence of Daily's book *Nature's Services* will be clear below, and it was Tilman's work on the effects of diversity that convinced me of the value of learning more ecology.

[2] The definition that I offer is close to the definition in J. P. Holdren, G. C. Daily, and P. R. Ehrlich (1995), and is the one used in G. M. Heal (1998). The Brundtland Report defines sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs." While eloquent, this is very thin on operational content. See World Commission on Environment and Development (1987). Sands suggested that as used legally, sustainability involves four components.

people think of current fisheries policies as unsustainable: clearly they are so according to this definition. Energy policies, involving the emission of greenhouse gases, imply an unacceptable risk of climate change, and are unsustainable both in common parlance and by this definition. So are important aspects of agricultural policies, those that result in loss of soil fertility, or in massive run-offs of nutrients into rivers, lakes, and ground waters.[3]

Given a working definition, a natural next step from an economic perspective is to ask: Do existing economic and political institutions lead us to make sustainable choices? This question is clearly rhetorical: the examples cited above show that the answer is "no." Two more questions follow immediately from this: Why not? And then: What can we do about this?

I think we can group the economic answers to the first question "Why not?" under three general headings. The first heading is that no part of our economic system explicitly recognizes the great value of many environmental systems: there are no economic decision-makers who are conscious of the importance of natural ecosystems to modern societies. The second—and this is closely related to the first heading—is that the economic incentives are wrong. This is tied to the public good nature of many environmental services. Finally, we place relatively little value on the future, and many of the benefits from environmental conservation, from sustainable economic and social organization, are far into the future. Changes in the economic, political, and legal framework, changes that in some cases are already under way, can make a significant contribution to the first two problems here. The third is more complex: better use of markets can certainly help, but a complete solution will require more far-reaching changes.

The Importance of Nature

How important is nature to modern societies? Because of our technological sophistication, we can easily come to see ourselves as remote from and independent of nature. This attitude is wrong and dangerous, and contributes very directly to the environmental problems that we face. The distinguished biologist E. O. Wilson describes our society as dependent on "prostheses" or artificial limbs to replace many of the functions historically performed by natural systems, a dependence that eventually brings many risks. Natural ecosystems provide critical infrastructure for all human societies. Societies derive a wide array of important economic and life-support benefits from biodiversity and the natural ecosystems in which it exists.[4] Biologists use the term "ecosystem services" to refer to these benefits. Daily

One is a commitment to preserve natural resources for the benefit of present and future generations. A second refers to "prudent," "rational," "wise," or "appropriate" use of natural resources. A third is an equitable use of resources, and a fourth implies that environmental objectives are integrated into development plans, and vice versa. See Philippe Sands (1997).

[3] For more detail, see Vitousek et al. (1997).

[4] This discussion draws heavily on, and at some points quotes, G. C. Daily (1997).

defined them as follows: "Ecosystem services are the conditions and processes through which natural ecosystems, and the species that are a part of them, sustain and fulfill human life. These services yield ecosystem goods, such as seafood, wild game, forage, timber, biomass fuels, natural fibers, many pharmaceuticals, industrial products, and their precursors. The harvest and trade of these goods represent an important and familiar part of the economy. Natural ecosystems also perform less appreciated but critical life-support services, upon which the well-being of all societies depends." Daily[5] listed these as:

- purification of air and water
- mitigation of droughts and floods
- generation and preservation of soils and renewal of their fertility
- detoxification and decomposition of wastes
- pollination of crops and natural vegetation
- dispersal of seeds
- cycling and movement of nutrients
- control of the vast majority of potential agricultural pests
- maintenance of biodiversity
- protection of coastal shores from erosion by waves
- protection from the sun's harmful ultraviolet rays
- stabilization of the climate
- moderation of weather extremes and their impacts
- provision of aesthetic beauty and intellectual stimulation that lift the human spirit.

As Daily explained, this array of services is generated by a complex interplay of biological, geological, and chemical cycles driven by solar energy and operating across a wide range of spatial and temporal scales. Soil fertility, for instance, is a product both of bacteria, whose fleeting lives may take place in a space smaller than the period at the end of this sentence, as well as of the aeonic, planet-wide cycles of major chemical elements such as carbon and nitrogen. Pest control is created by both natural enemies (e.g., birds; bats; parasitic wasps, ladybugs, spiders, and other predacious arthropods; fungi; viruses) and by climate patterns generated globally. The stratospheric ozone layer that shields Earth's surface from ultraviolet radiation was originally produced primarily by the photosynthetic activities of blue-green algae and by photochemical reactions occurring high in the atmosphere.[6] Ecosystem services operate on such a grand scale and in such intricate and little-explored ways that most could not be replaced by technology.[7] Above I listed 14 important ecosystem services. Some of these could be replaced in part by synthetic alternatives, while for the great majority not even a partial replace-

[5] Daily, op cit.
[6] These examples are taken verbatim from Daily (1997), which contains many others.
[7] For more details see Daily (1997).

ment is currently possible. We have no substitutes for many critical services provided by the natural world: we are therefore very dependent on nature but alarmingly unaware of this and fail to capture this dependence in our economic decisions.

Incentives and Markets

What does it mean to say that the economic incentives are wrong? A concrete case will make the point. Consider conservation of Amazonian rainforests. This has some costs and some benefits: I believe, as do most people who have thought carefully about this, that the benefits greatly outweigh the costs. The costs are borne by people who live in and around the forest: they are opportunity costs, costs of not being able to use the land for cash crops or for ranching, and of not being able to sell the timber. Some of the benefits, on the other hand, accrue to everyone: to you, to me, to all humans, because they are in the form of carbon sequestration and biodiversity support. Others accrue to people in Brazil, but not to those in the forest region: these are benefits from stream and flood control, regional climate control, ecotourism, and many others. So in essence one set of people pay the bill and another set eat the meal. This is obviously not a stable situation: the bill payers will drop out. We can only expect a forest to be conserved if some of benefits from conservation accrue to the bill payers. The essence of this rainforest example could be replicated many times.

How can we change institutions to avoid this problem? Four examples will provide much of the groundwork for arguments about how more effective use of markets can realign incentives and at the same time lead to a more realistic appreciation of the value of natural ecosystems.

Growing carbon and the Kyoto Protocol

The Kyoto Protocol could provide an interesting example of how markets might provide powerful incentives for environmental conservation. The final version of the Protocol may contain provisions for carbon sequestration credits: under these provisions, countries that remove carbon from the atmosphere, for example by growing trees, will receive credits for this in the form of tradable greenhouse gas emission permits. No one knows what exactly the value of these permits will be. Preliminary economic calculations[8] suggest that their value could be in the range $15 to $100 per ton of carbon or equivalent. What would this mean for the economics of conserving tropical forests? Again, we don't know for sure, but we can do some rough calculations. Growing moist tropical forests remove carbon from the air at a rate in the range of 5 to 15 tons per hectare per year, possibly more.[9]

[8] The range of $10 to $50 emerges from simulations conducted by the Program on Information and Resources at Columbia University using a modified version of the OECD's GREEN computer model of the global economy. W. D. Nordhaus of Yale has reported estaimates as high as $100 (pers. comm.).

[9] Personal communication, Steve Pacala, Ecology Department, Princeton.

Taking these two ranges of numbers together, we see that growing forests could be remunerated by carbon sequestration credits at a rate of $75 to $1500 per hectare per year. This is a lot of money: ranches in Costa Rica, for example, make profits of at most $100 to $125 per hectare per year. Reforesting has a one-off cost of planting the seedlings, which can be as high as $900 per hectare.[10] Even with this initial cost, it seems possible that if the world as a whole pays for just one of the many services of tropical forests, then this could change radically the economics of forest conservation. On a very small scale, some of this is already happening through the schemes for joint implementation encouraged by the Global Environment Facility and the World Bank.

Analytically this example demonstrates the following point. The sequestration of carbon by forests is a global public good. By stabilizing the climate it benefits all of humankind. Typically it is difficult for the providers of such a service to appropriate all or even a significant part of the benefits, so that it is under-provided, as we noted above. A combination of a tradable permit system for greenhouse gas emissions plus a regime of giving permits for sequestration may allow providers of sequestration services to capture the full economic value of what they provide.[11]

Watersheds

Another example is already up and operating. Ninety percent of New York's water comes from a watershed in the Catskill Mountains. Until recently a purification process carried out by roots and microorganisms in the soil as the water percolates through, together with filtration and sedimentation occurring during this flow, were sufficient to cleanse the water to U.S. Environmental Protection Agency standards. Recently sewage, fertilizer, and pesticides in the soil reduced the efficacy of this process, to the point where New York's water no longer met EPA standards. The city was faced with a choice: restore the integrity of the Catskill ecosystems, or else build a filtration plant at a capital cost of $6–8 billion, plus running costs of the order of $300 million annually. In other words, New York had to invest in environmental conservation or in treatment facilities.

Which was more attractive? Investment in conservation in this case meant buying land in and around the watershed so that its use could be restricted, subsidizing the construction of better sewage treatment plants, and buying conservation easements in the region. The total cost of measures of this type needed to restore the watershed is expected to be in the range of $1–1.5 billion. So investing $1–1.5 billion in conservation could save an investment of $6–8 billion in treatment facilities, giving an internal rate of return of between 90% and 170%.[12] This return is an order of magnitude higher than is normally available, particularly on relatively

[10] Personal communication, Daniel Botkin.

[11] The assertion that markets give appropriate incentives for providing public goods is a surprising one to most economists in view of the free-rider problem. For a detailed analysis, see G. Chichilnisky and G. M. Heal (1998a).

[12] The discussion of the New York watershed case is taken from G. Chichilnisky and G. M. Heal (1998).

riskless investments. These calculations are conservative, as they consider only one watershed service, although watersheds, typically forests, often provide other important services.

In 1997 New York City floated an "environmental bond issue," and will use the proceeds to restore the functioning of the watershed ecosystems responsible for water purification. The savings produced will meet the cost of the bond issue: the savings are the avoidance of a capital investment of $6–8 billion, plus the $300 million annual running costs of the plant. The cash that would otherwise have gone to these will pay the interest on the bonds.

The market could have handled this transaction, by using the technique of securitization. This involves issuing tradable contracts, securities, entitling the owners to a fraction of the benefits from a venture. In this case, the securities would entitle their owners to a fraction of the cost savings resulting from watershed restoration: they would be "watershed securities." In exchange for the securities, investors would contribute the capital needed for restoration. In effect they would invest in the restoration and receive in return a share of the benefits. Securitization is a technique already extensively used for attracting investors into a venture. This financial structure is already used in securitizing the savings from increased energy efficiency in buildings. Securitization of the savings involves issuing securities entitling their owners to a specified fraction of the savings. Typically these contracts are tradable, issued to the providers of capital, and can be sold by them, even before the savings are realized. This is a way of making investment in saving energy attractive to the investing public and institutions: it does not imply any transfer of ownership of the underlying asset. The U.S. Department of Energy has a standard protocol for estimating the savings from enhanced building energy efficiency and several financial agencies are willing to accept these estimates of energy savings as collateral for loans.

In the New York watershed case, the city could have opened a "watershed savings account" into which it paid a fraction of the costs avoided by not having to build and run a filtration plant, which would pay investors for the use of their capital. The purpose of securitization is to make it possible to finance projects such as New York's watershed restoration without using the credit of the City itself, an important issue in developing countries whose metropolitan areas often do not have credit ratings comparable to New York's.

One could take the introduction of market forces a step further. Imagine a corporation managing the restoration of New York's watershed. It has the right to sell to New York City the services of the ecosystem, the provision of water meeting EPA standards. Ownership of this right would enable it to raise capital from capital markets, to be used for meeting the costs of conserving New York's watershed. Of course some regulation would be needed: for example, the corporation would be a natural monopoly so that it would be appropriate to regulate its prices. It would also be reasonable to place some restrictions on the modifications that it could make to the natural ecosystems in the watershed area.

Analytically, what is the general point illustrated by this example? It is that water is a good for which individuals or municipalities are willing to pay. They are willing to pay for quality as well as quantity, and this in effect puts a price on the water management and purification services provided by a watershed. It generates an indirect demand, what economists call a derived demand, for watersheds. In this case there is a possible replacement for the ecosystem services provided by the watershed: this is the filtration plant. The cost of this, in the New York case $8 billion plus, puts an upper limit on what it would make sense to spend on restoring the watershed, looking at it only from the water management and purification perspective. Of course, the watershed may have many other values to society: in this case, the Catskills are a much-valued recreational area, and society has a substantial willingness-to-pay for this in addition to the watershed services. An important aspect of this story is that by improving sewage systems in the Catskills and other measures to reduce pollution there, and by buying conservation easements, the city of New York has improved the quality of life and injected a considerable amount of income into the Catskills community. It has therefore provided some financial compensation to the residents of the area to be conserved, and given them a direct financial stake in the conservation.

Ecotourism

Another powerful example comes from South Africa, in the form of the Conservation Corporation, or Conscorp, a company started as a private venture and recently floated on the London Stock Exchange.[13] This imaginative venture has capitalized on the demand for ecotourism and hunting: this demand is such that land yielding $25 per hectare annually for ranching and $70 per hectare in cropping can yield between $200 and $300 per hectare as part of a reserve managed for tourism or hunting. Conscorp contracts with landowners to incorporate their land in its reserves: it does not buy the land outright. Landowners have to maintain their land in accordance with tightly specified regulations and to stock it with specified animals. Conscorp manages the business part of the operation, bringing tourists and hunters, building facilities and providing guides and vehicles. To date they have restored several hundred thousand hectares of farmland to their original ecosystems. An interesting detail is that the presence of lions will add about 30% to the revenues from an area, so that the incentive to restock with these is great.[14] Supporting lions, at the top of the food chain, requires that most of the rest of the chain beneath them be there: what the lions eat, what the lions' food eats, and so on. There is a strong economic incentive to do a thorough job of restoration. This is in fact clear from the very explicit

[13] Their web site is http://www.world-travel-net.co.uk/conscorp. For more discussion see http://economics.iucn.org. For more on Conscorp see Terry Anderson, "Enviro-Capitalists: Doing Good While Doing Well." Published by Rowman and Littlefield, 1997.

[14] Personal communication, Craig Packer.

rules used by the Conscorp, whose Articles of Association for a specific reserve state that its aims are "to promote and conserve endemic wildlife within the confines of the area...; to establish the Reserve as a sanctuary in perpetuity for endemic wildlife and habitat so as to ensure sustainable resource utilization...; to endeavor to increase the area of the Reserve; and to maximize the long term economic and ecological value of the properties...." Landowners even agree not to keep any domestic animals, dogs and cats included. An interesting quote from a South African writer captures some of what is happening in this movement:

> "The interesting thing is that untold hundreds of thousands of hectares and morgen that even a few years ago were scrub grazing for a mixture of game and cattle have now been entirely allocated to game. Why? Economics, as always. Game pays its own way, eats nearly anything, is more resistant to disease and predators and generally produces a higher and better use for the land.... Even the old enemies become assets to the farmer who switches from cattle to game. One friend of mine used to lose as many as thirty calves a season to leopards.... Now those same leopards are worth a cool $3000 to $4000 to sport hunters, not a bad trade-off for animals that caused a liability of well over ten grand and had to be poisoned! Tell me, is that bad for leopards?"[15]

Similar developments are occurring in Kenya. The land there is less productive and the country has less infrastructure, so that numbers are all lower. But they tell the same story in terms of incentives: in the Laikipia region of Kenya ecotourism can bring $5 to $30 per hectare per year, compared with less than $2 per hectare per year for traditional livestock husbandry.[16]

Prospecting for pharmaceuticals

Bioprospecting is another activity that can yield cash for conservation. Bioprospecting means seeking for leads in the development of new drugs, or new chemicals for use in agriculture, by looking at biological resources. As a matter of fact, over 60% by value of prescription drugs in the U.S. are or were initially derived from plants and insects, so this is a reasonable place to start looking. The key point is that certain plants and animals are known to produce substances that are highly active pharmacologically. Plants that live in insect-infested areas produce substances that are poisonous to insects, and these have been used as the basis for insecticides. Some snakes produce venom that paralyzes parts of the nervous system, and others produce venom that reduces blood pressure. Other insects produce anti-coagulants. All of these have been adapted for medical use. Observations of this type have led most major drug companies to pursue bioprospecting as a way of finding

[15] Capstick, cited by Anderson, op cit. note 9.
[16] D. I. Rubenstein (1993).

new pharmacologically active substances to serve as a basis for drug development. Typically they have sought these compounds in the tropics, in areas where there is extensive inter-species competition, or in other extreme areas. They have been willing to pay quite substantial sums for access to these regions, and have made deals with host countries that involve giving them a royalty on the products that might eventually be based on this prospecting. Such royalties could be large relative to the incomes of the countries concerned. Two concrete examples will illustrate the potential salience of this point. The key enzyme in the polymerase chain reaction (PCR), a reaction central to much modern genetic testing and indeed to a broader range of biotechnologies, was discovered in one of the hot springs at Yellowstone National Park. There it evolved the resistance to heat that is critical to its role in PCR. Now virtually every biotechnology laboratory uses derivatives of this enzyme, whose commercial value is immense. Merck Inc., one of the largest pharmaceutical companies in the U.S., has an agreement with a Costa Rican agency called InBio (Institute Nacional de La Biodiversidad) for bioprospecting rights in Costa Rica. The terms of the agreement are that Merck paid InBio a fixed sum, to be used for forest conservation, in exchange for the right to receive samples collected by InBio and to use these as the basis for new product development. Should any of them prove commercially successful, Merck will pay InBio a royalty on the revenues generated. Similar agreements are in place between other U.S. pharmaceutical companies and other regions of Central and South America. There has been some controversy about the potential value of bioprospecting to developing countries. The discovery of the enzyme for the PCR reaction, the agreement between Merck and InBio and several other drug discoveries based on plants from developing countries, led to a wave of optimism, some perhaps excessive, about the potential commercial value of in situ biodiversity in developing countries. After a reaction against this, a more balanced position is emerging, although we still lack extensive practical experience. Real data will emerge only slowly, as the development and testing of drugs is a slow process, taking at least 10 years. However, recent calculations have suggested that in some of the world's biodiversity hotspots, the right to bioprospect may be worth as much as $9000 per hectare, about a century of ranching income.[17] The key insight in these calculations is that prior knowledge of the nature of the ecosystems in a location can improve estimates of the probability of finding commercially interesting compounds there. In practical terms developing countries can clarify the commercial attractions of their biodiversity by research on the ecosystems of which it is a part. This is rather like a country with potential oil reserves engaging in basic geological prospecting before seeking to negotiate leases for oil development. The results may be positive or negative, but either way they will give it a better view of its prospects.

Here the economic point is recognizing the value of the knowledge that can be derived from natural systems, and then establishing intellectual property rights in

[17] See G. C. Rausser and A. A. Small (1998). These figures apply only to selected locations recognized as rich in biodiversity and should not be taken as typical of tropical regions.

that knowledge. The property rights are needed to ensure that some of the value ultimately created by that knowledge returns to the country from which it is derived.

These examples—carbon sequestration and the Kyoto Protocol, the New York watershed, the Conscorp and bioprospecting—make two points. One is that, with the right institutional structures and property rights in place, the market can be used to attach a value to certain natural assets. The second is that, if this is done, the incentive problem can be corrected. If the market will yield a higher return through conservation than through any other use, then entrepreneurs will find a way to conserve. Of course, I have cited particular examples: they do not imply that all valuable natural capital can be conserved this way. What they do suggest is that we should look more into the potential of this approach. Preliminary estimates suggest that up to 10% of the land area of the U.S., and a comparable or greater area worldwide, could be economically conserved on the grounds of watershed protection.[18] I am not aware of comparable studies for ecotourism, but this has certainly become a major industry in several regions of the world, including Central America and east and southern Africa. In southern Africa, ecotourism has already made a substantial impact on the conservation of many animals. About 18% of the total land area of Angola, Namibia, South Africa, Botswana, Zimbabwe, and Mozambique is now devoted to national parks or to game ranching, the term used to describe commercial use of land to support natural flora and fauna for tourism. Some countries in these regions are now earning as much as one-third of their foreign exchange from ecotourism.

A final observation: the different ways of obtaining a return described in the examples above are not mutually exclusive. A forest could obtain returns from carbon sequestration, bioprospecting, managing a watershed, and ecotourism. In fact, the region of the Mata Atlantica (Brazilian coastal rainforest) inland from Rio de Janeiro is in a position to do exactly this. It manages the watershed for Rio in much the way that the Catskills region does for New York. It also manages the streamflow of the river Rio Paraibo do Sul, which provides most of the hydropower for Rio. These two services make it truly a major utility for Rio, with great economic value. Additionally, it supports a wide range of endemic species, sequesters carbon, and acts as a magnet for tourists. Currently the region obtains a financial return only on one of these activities, ecotourism.

The Role of the Market

What are the general points to emerge from these examples? Obviously they are encouraging, and suggest the possibility of generating real economic incentives for conservation in several important cases. Which services of the environment are amenable to management by the market? One clear prerequisite is that the ecosystem to be conserved must provide goods or services to which a commercial

[18] These numbers come from W. V. Reid (1998). In this paper, Reid began to explore the scope for generalizing the New York case to other regions.

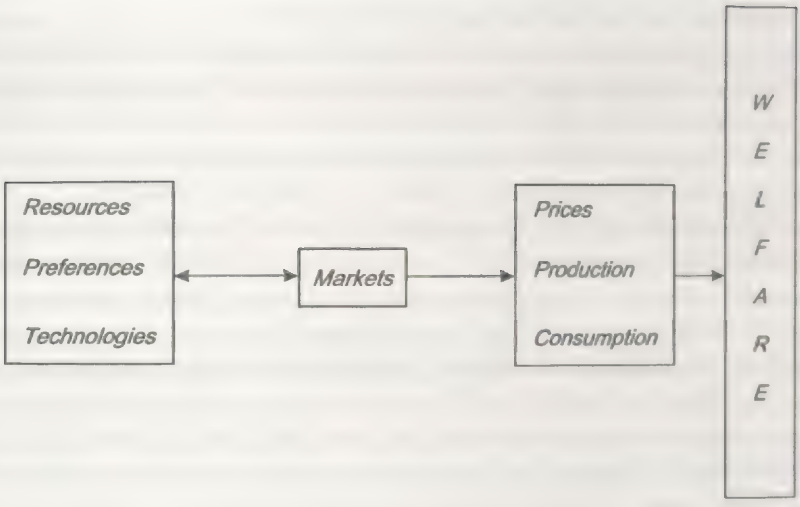


Diagram 1.

value can be attached. Watersheds satisfy this criterion: drinkable water is becoming increasingly scarce, and indeed the availability of such water is one of the main constraints on health improvements in many poorer countries.

Commercial value of an ecosystem service is necessary but not sufficient for privatization: some of that value has to be appropriable by the producer. A critical issue in deciding whether ecosystem services can be privatized is the extent to which they are public goods. Pure public goods are challenging to privatize: they are goods which if provided for one are provided for all. It is hard, though often not impossible, to exclude from benefiting from their provision those who do not contribute to their costs, so that their providers cannot appropriate all of their returns. Markets typically do not manage such goods well. Water quality is a public good, in the sense that if it is improved for one user of a watershed, then it is improved for all. But the consumption of water itself is excludable, so the watershed case involves bundling a public with a private good. Ecotourism is also a hybrid: the existence of viable populations of flora and fauna is a public good from which everyone benefits to some degree. But this is not what game ranches are selling: they are selling room-nights in game lodges, meals, the services of guides, and many other accoutrements of up-market tourism. These are private goods. So in both of these cases conservation conserves a public good (water quality, stream flow, wildlife), but the incentive to do this comes from the resulting enhanced ability to sell an associated private good. The presence of a public good means that the market incentives to conserve are less strong than required for overall economic efficiency, so that a fully efficient outcome will not be realized, but by capturing some of the value the market improves on the pre-existing situation. The same is true of bioprospecting, although in this case there is the additional interesting point that the final product, knowledge, has many of the features of a public good, and so has to be protected by intellectual property rights regimes.

Carbon sequestration is different. Here the market is paying for a pure public good. Using markets in this way is novel: the first significant initiative was in the U.S. 1990 Clean Air Act, which introduced a system of tradable permits for controlling the emission of sulfur dioxide. In this case the conditions for overall economic efficiency depend on the initial distribution of abatement rights, leading to an interesting connection between efficiency and distribution.[19]

We can indicate the role of the market in a diagram. Diagram 1 represents the role of the market in what economics texts portray as a perfect world: the basic data of the economy are its resources, preferences, and endowments. These interact in a market to determine what is produced, the prices of goods and services, and the pattern of consumption. These economic variables, production prices and consumption, determine society's economic well-being. The market plays a complex role: it determines which resources are converted by technologies to goods and services that satisfy people's preferences: it also manages the resource base, in the sense that if a resource is scarce its price rises, reducing use and providing incentives to search for more.

In fact, because many ecosystem services are not mediated by a market, the situation we face is more like Diagram 2:

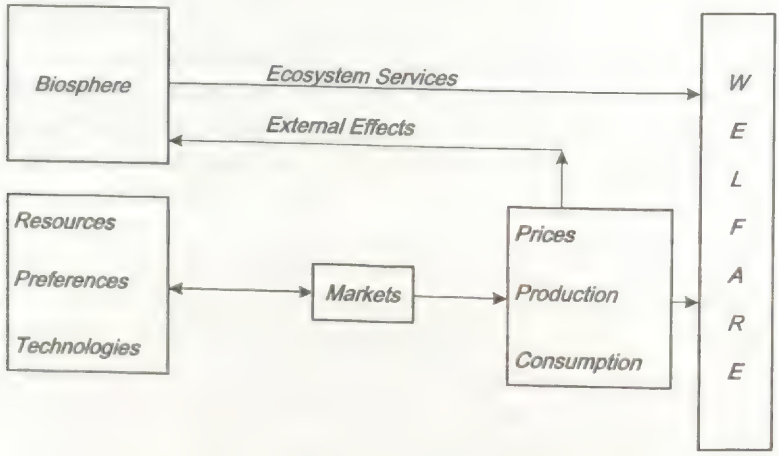


Diagram 2.

This shows the biosphere having a direct impact on human welfare, not mediated by a market, and also human economic activities affecting the biosphere externally to the market. This is the route by which most environmental damage occurs: as a byproduct of market activity, as with pollution or deforestation. Emissions of greenhouse gases and ozone-depleting gases are classic examples of production processes and consumption patterns having effects on the biosphere that are external to the market, and the Kyoto Protocol, as we saw above, is an attempt to bring them within a more broadly defined market. The examples reviewed earlier all portray aspects of ecosystem services that can be brought under the control of

[19] See G. Chichilnisky and G. M. Heal (1998).

the market, and make the point that there are many such cases. But there are also important aspects of ecosystems that cannot be effectively safeguarded by the market: for these we will need to develop non-market institutions if we are to conserve them effectively. For example, macaws are threatened by two separate forces: their habitat is being destroyed, and they are being captured for the pet trade. Conservation of their habitat, via incentives for carbon sequestration or via ecotourism, is necessary but not sufficient for their preservation. Only regulations such as the endangered species act or CITES (Convention on International Trade in Endangered Species), can complete the picture and ensure their survival. Alternatively there may be systems or species that have little economic importance, but are of great ethical or cultural significance, such as the bald eagle in the U.S., and other cultural or national symbols. In the light of such cases, we must ultimately develop a framework like that in Diagram 3, with human interaction with some aspects of the biosphere mediated by markets and with other aspects controlled by quite different institutions.

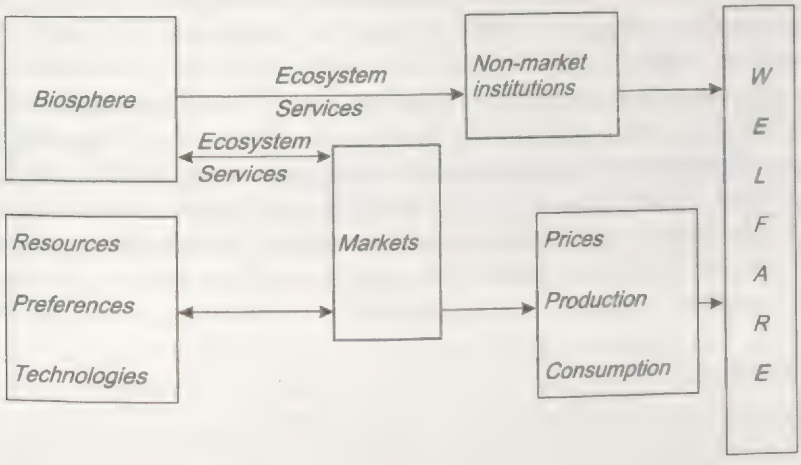


Diagram 3.

National Income

A lot has been written on the shortcomings of national income as a measure of economic well-being, particularly in the context of significant environmental impacts. The central issue here is that many environmental services are, as we have seen, provided outside of the market, so that their contribution is not registered by national income. More specifically, a decline in these services is therefore not reflected in a decline in national income. In fact, to return to the example of the Catskills watershed cited above, if New York City had destroyed the watershed totally, and replaced it by an expensive filtration plant, then national income would have risen, probably by more than it actually did as a result of conservation of the watershed. To the extent that environmental contributions are brought within the ambit of the market, the severity of this problem is reduced and the accuracy of national income is improved.

Valuing the Future

Valuing the future is critical to sustainability. Environmental assets provide flows of services over long periods of time. New York's Catskills watershed has purified water and controlled stream flow for hundreds if not thousands of years, and if left intact will continue to do so for at least as long again. Insects have pollinated flowers for much, much longer, and could continue as long again if not driven extinct by pesticides.[20] No human systems have such life spans. Physical assets produced by humans last perhaps 10, perhaps 50, years: at best their life spans are measured in decades rather than centuries or millennia. Knowledge and culture are the only assets we produce that can rival natural assets in their durability: Shakespeare thrives at half a millenium, and Plato and Pythagorus at several millennia. Because of the totally different time scales of the capital assets that humans and nature produce, the techniques that we use for valuing our capital assets really cannot be applied to natural assets. We value the capital that we produce by cost-benefit analysis, taking as its benefits the present discounted value of the flow of services that it produces. By discounting at rates in the range of 5% to 10%, we in effect choose an implicit time horizon, a date beyond which nothing matters. This is in the region of 20 to 30 years: $e^{-0.05X20} = 0.22$ and $e^{-0.1X20} = 0.05$ so that at a 5% discount rate one dollar 20 years hence has a present value of 22 cents and at 10% its value is 5 cents. When we apply this kind of calculation to environmental assets, we are cutting out most of the contributions that they will make to human societies: we are taking account of 20 years of the contributions of assets that could contribute, at no extra cost, for 20 decades or perhaps 20 centuries. Clearly we are undervaluing them grossly.

Discounting

Discounting has always been a source of controversy between economists and those from other disciplines interested in the environment. After all, if you discount at 5% over 100 years, then you are giving a future dollar a present equivalent of only e^{-5} = \$6.7379 times 10^{-3}, which is roughly 2/3 of a cent. It is hard for a non-economist to reconcile this with taking the future at all seriously. Perhaps less well known is the fact that discounting has also been a source of controversy within the economics profession. Frank Ramsey, the first person to think seriously about dynamic economics and author of a seminal 1926 article on long-run planning, commented that discounting "is ethically indefensible and arises merely from the weakness of the imagination."[21] His contemporary Roy Harrod added that it is a "polite expression for rapacity and the conquest of reason by passion."[22] These are strong words, from people who clearly believe that they are right.

[20] See Nabhan and Buchmann (1997).

[21] F. Ramsey (1926). If you sense a touch of ecclesiastical certainty in this remark, you would be correct: although Frank Ramsey died young, his brother became the head of the Church of England. Ramsey was an interesting character and a friend and colleague of John Maynard Keynes.

[22] R. Harrod (1948).

Why has discounting been controversial in economics, and why has the controversy been particularly acute in the environmental area? The key point about the environmental area is that it often forces one to consider long time horizons. In the climate change area, a century is the minimum time horizon to make sense. Probably it is too short. The same is true about species extinction and biodiversity loss and disposal of nuclear waste. Scientific processes relating to the environment naturally unfold over this type of horizon. Such time horizons are completely outside the normal range in economic decision-making. Corporations and governments normally look at most decades ahead, rather then centuries. The longest "normal" time horizons in economics are those for infrastructure investments such as power stations, where 30 years is a possible horizon. Five to 15 years is much more common as a planning horizon, and is still longer than the average. Horizons of this type are not long enough to raise one of the issues central to long-term environmental problems, namely equity between generations. For 5 to 15 years in the context of business plans, the efficient use of money is the central issue. For a century or more, when considering the planet's life support systems, equity and "sustainability" naturally come to the center of the stage (although efficient use of capital does not leave the stage). So normal intertemporal economic problems and long-term environmental problems have quite different time scales and involve different issues. It is natural that the methodology for one does not fit the other perfectly.

In resolving long-term environmental issues, we want to achieve two aims: one is to strike a fair balance between the present and the future, and the other is to use our limited economic resources, including capital, efficiently. What is involved in efficient use of capital? Efficient use of capital requires that the rates of return on all types of capital and durable goods and equipment (including durable natural systems) are the same. Establishing markets in all such goods, or in their services, will meet this condition, which in a market context becomes a no-arbitrage condition. So we should let a market charge for the use of capital, which gives rise naturally to a cost of capital. We need a cost for capital to make sure we don't waste it: however green your complexion, you certainly want to use society's capital as productively as possible. However, establishing a market price for the use of capital and discounting the future are two quite different matters. Neither implies the other.[23]

How do we strike a proper balance between present and future? This is a harder question to answer: in the end, this is a matter of ethical judgment, the same category of judgment as we make when we say that a particular distribution of income is acceptable, or is too unequal. This does not mean that it is just a matter of opinion, and that all opinions are equally good. There are some basic principles that we can establish, and these can provide valuable guides. But in the end, there is no purely scientific definition of a "proper" balance between present and future. The controversy within economics over discounting arises from disagree-

[23] See, e.g., P. S. Dasgupta and G. M. Heal (1979: chapter 7).

ments about what is a proper balance, plus perhaps some confusion of the issues of intergenerational equity and efficient use of capital and a lack of clarity on how they interact.

What about adopting a more future-friendly decision-making algorithm than exponential discounting? Certainly there are possibilities here.[24] There are approaches to valuing income streams over long time periods that give more weight to the future than the standard practice, and that are just as compelling and logical, indeed from many perspectives even more so. In fact, empirical studies seem to suggest that individuals do not follow the standard model when making choices over time. It appears that most individuals use discount rates that are lower, the longer the time horizon that they consider.[25]

Where does this leave us with respect to valuing the future, particularly the long-term future? It tells us that there is nothing particularly sacrosanct logically about the usual way of doing things: equally compelling alternatives are available, and will place an order of magnitude more weight on the future. Most of the alternatives are consistent with the competitive market, in the sense that they imply intertemporal price sequences at which maximizing behavior will support socially optimal outcomes. However, maximizing behavior may not mean maximizing the present value of profits: it may mean maximizing the long-run level of profits or some combination of this and a present value of profits. Intuitively this makes sense: fishing fleets are clearly seeking to maximize short-run profits to the detriment of long-run profits, and paying more attention to the latter would lead to quite different behavior. Unfortunately it is not clear what it would take to change these behavior patterns throughout most of society. Some points are clear, however. Certain types of institutions make it their business to lobby for future-oriented decisions, and to try to tip the scales in that direction. These are the NGOs (Non-Governmental Organizations). In particular, in the U.K., the National Trust has as its primary purpose the purchase of land and properties of outstanding environmental or historical importance, to be held in trust for the future. The Nature Conservancy in the U.S. has a similar role vis-à-vis the environment. More should be done, but these institutions indicate a potential mechanism for implementing future-oriented decisions even in a society where most decisions involve short time horizons.

Another important general point is that this is an area where we cannot expect to rely on market forces without a lot of intervention and assistance. There are no markets for goods and services well into the future, so that with respect to

[24] They are discussed at length in Heal (1998). An interesting possibility is to be found in G. Chichilnisky (1996).

[25] One rationalization of this is that they respond to proportional rather then absolute changes in distance in time. Postponing a reward by one year from the first to the second year, and by one year from the twentieth to the twenty-first, are both one-year postponements, but one matters a lot and the other doesn't. One is a doubling and the other a 5% change. If it is the percentage change that matters and not the number of years, then people discount according to the logarithm of time, thus: e-0.1 log t for a 10% discount rate. This leads to a far higher relative value on the future. For example, after 100 years discounting logarithmically at 10% gives almost 14,000 times more weight than discounting exponentially. After 50 years, the ratio is 100. Over only 10 years, the difference is a factor of two.

the future the market cannot perform effectively its role of bringing together suppliers and consumers and providing incentives for the former to meet the needs of the latter. There are actually some futures markets, markets for goods to be delivered in the future, but these are for trading specialist financial instruments and cover only a few years ahead. Even if we had markets for goods 50 years ahead, they would not solve our problem because future generations, the people whose welfare concerns us, would not be available to trade on these markets. The markets could therefore not reflect their interests. Market forces do determine current levels of investment, which of course affect future levels of welfare, but they usually do this with reference to short-term considerations. It would in fact be a remarkable coincidence if they were to provide the right incentives to build for the long-run future.

Having said this, I should in fairness note that some generations have endowed their successors generously. The Victorians endowed subsequent generations lavishly with many types of infrastructure: to take a New York example again, they built a water distribution system that has been adequate until very recently. However, other generations have behaved in quite the opposite fashion: think of our generation with respect to fish stocks, the Romans with respect to the natural productivity of the Sahara (which was the "breadbasket of Rome"), or medieval Britain with respect to forests. It is suggestive that most of the examples of parsimony with respect to bequests relate to natural resources, and those of generosity relate to manufactured resources.

Liquidity constraints and capital markets

Existing financial markets can in some cases be useful in conservation, without any changes. Suppose that a developing country has valuable environmental assets, such as forests, and can derive some income from these, for example by carbon sequestration or bioprospecting. But suppose in addition that this country is very poor, and is in a state of economic and financial crisis, in desperate need of income. Many countries meet the assumption of being desperately poor, but unfortunately not the assumption of deriving income from carbon sequestration or biodiversity: Indonesia is very poor and in urgent need of income. Sadly it is not yet able to generate income from the services that its unique and extensive biosphere provides to the rest of humanity. Our hypothetical country faces a steady stream of income continuing into the future from its environmental assets, but has an acute need for income now. It would gladly trade future income for present income, and the inability to do so might lead it to choose the strategy of selling forests as timber, which gives lots of cash now and none in the future, rather that conserving them, which gives less cash now but more in total. Here financial markets can help: one of their main purposes is to move income over time, and let people and institutions spend now in anticipation of future income. This is exactly what a mortgage does: lets the borrowers consume in excess of income now in exchange for consuming less than income in the future.

In a situation such as the one we have described, financial institutions or markets could readily provide current income against the security of repayment from future environmental income. To put flesh on these bones, consider the case of Costa Rica and Merck. Costa Rica's InBio signed an agreement with Merck under which it provides Merck with specimens to be tested for pharmaceutical potential, in exchange for a one-time payment and a royalty on any drug eventually produced. The right to a royalty is a right to a part of the profits of specified Merck products. This right could eventually produce huge payments, but these may be 20 or more years in the future: after all, it can take 15 years to take a drug from conception to market. Costa Rica has here a right to an uncertain future income. Some or all of this right could be sold for cash now: it constitutes a form of security deriving its value from profits on certain Merck products. If Costa Rica were to suffer from an acute cash crisis, this ability to convert the agreement into cash now would enable it to bring future income forward and avoid pressure to liquidate environmental assets because of cash constraints.

The use of financial markets to anticipate income does not change fundamentally the way we value the future relative to the present. It allows us to avoid situations where current income shortages might force us into shortsighted choices, which we will subsequently regret. A current income crisis can push the value of the future into insignificance relative to the present. The ability to anticipate future income can avoid this.

Conclusions

To summarize: to date, markets have not performed notably well in conserving our planet's environment. Indeed, they have done quite the opposite. But this is not intrinsic in markets. They can be reoriented in a positive direction, in which case their potential for good is immense. They will not solve all problems, but they will reduce the need for governmental action and focus it on a much smaller number of cases than is currently the case. Probably the biggest shortcoming of markets is in their treatment of the future, and this is also likely to be one of the hardest to overcome.

References

1. Holdren, J. P., G. C. Daily & P. R. Ehrlich. 1995. The meaning of sustainability: Biogeophysical aspects. *In* M. Munasinghe & W. Shearer (editors), Defining and Measuring Sustainability: Biogeophysical Aspects. World Bank, Washington, D.C.
2. Heal, G. M. 1998. Valuing the Future: Economic Theory and Sustainability. Columbia Univ. Press, New York.
3. World Commission on Environment and Development. 1987. Our Common Future. Oxford Univ. Press.
4. Sands, P. 1997. Environmental protection in the 21st Century: Sustainable development and international law. Paper presented at a symposium at La Pietra on Envi-

ronmental Law. In Environmental Law, the Economy and Sustainable Development. Cambridge Univ. Press. Editors Richard Revesz and Richard Stewart. Summer 2000.

5. Vitousek, P. et al. 1997. Human domination of Earth's ecosystems. Science 277: 494–499.
6. Daily, G. C. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.
7. Chichilnisky, G. & G. M. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–630.
8. Chichilnisky, G. & G. M. Heal (Editors). 1998a. Environmental Markets. Columbia Univ. Press, New York.
9. Rubenstein, D. I. 1993. Science and the pursuit of a sustainable world. Ecological Applications 3: 585–587.
10. Rausser, G. C. & A. A. Small. 1998. Valuing research leads: Bioprospecting and the conservation of genetic resources. Paper presented at the Conference on Managing Human-Dominated Ecosystems at the Missouri Botanical Garden, March 1998. *Journal of Political Economy*, 108, 1 (Feb. 2000), 173–206.
11. Reid, W. V. 1998. A business plan for ecosystem services: Extending the New York City Watershed Model to other geographic regions and other ecosystem services. Paper presented by the World Resources Institute at the Conference on Managing Human-Dominated Ecosystems at the Missouri Botanical Garden, March 1998.
12. Nabhan, G. P. & S. Buchmann. 1997. Services provided by pollinators. Chapter 8 of G. C. Daily (1997).
13. Ramsey, G. 1926. A Mathematical Theory of Saving. *Economic Journal*, (1928), 38: 543–559.
14. Harrod, R. 1948. Towards a Dynamic Economics. Macmillan, London.
15. Dasgupta, P. S. & G. M. Heal. 1979. Economic Theory and Exhaustible Resources. Cambridge Univ. Press.
16. Chichilnisky, G. 1996. An axiomatic approach to sustainable development. Social Choice and Welfare 13(2): 219–248.

A View from the South

The Environmental Impact of Globalization on Latin America: a Prospective Approach

Gilberto C. Gallopín
United Nations
Economic Commission for Latin America and the Caribbean (ECLAC)
Av. Dag Hammarskjold s/n
Santiago, Chile

Graciela Chichilnisky
Columbia University
535 West 116th Street
New York, New York 10027, U.S.A.

Abstract

Current changes in Latin America include the abandonment of the economic pattern of import substitution, a growing opening of the national economies, a continental wave of political democratization, an apparent economic recovery from the "lost decade" (the 1980s), a growing social polarization, a worsening of environmental problems, the growing influence of the market, and the most intense urbanization process on the planet. The aim of this paper is not to discuss the advantages or disadvantages of the prevailing economic pattern, but only to analyze some of the possible environmental implications derived from the way of insertion of the countries of the region in the global economy. The region as a whole is relatively well endowed in terms of natural resources. With little more than 8% of the global population, Latin America has 23% of the potentially arable land, 10% of the cultivated land, 17% of the pastures, 22% of the forests (and 52% of the tropical forests), and 31% of the permanently usable freshwater. It has not less than 3% of the world reserves of fossil fuel and 19% of the technically usable hydroelectric power.

Regarding economic globalization, the general argument from the environmental point of view is not that international trade is negative and that autarchy is desirable, but rather that a certain degree of regulation is necessary to reach a "sustainable free trade." The technological aspect of globalization is so important that it is possible to speak of a true techno-economic revolution or *Knowledge Revolution* (see also Chichilnisky's chapter in this book), led by microelectronics and the information technologies, and accompanied by a constellation of developments based on new technologies intensive in science (biotechnology, new materials, new energy sources, nanotechnology, etc.). From the point of view of their environmental implications, many of the new and emergent technologies exhibit interesting differences with the previous technological paradigm. The attributes of the new paradigm having higher strategic interest can be characterized as ambivalence, flexibility, and knowledge-intensivity. The *technical* potential for ecologically sustainable development is higher today than in any moment of the past. However, the direction toward which the trajectories of the new techno-economic paradigm seem to be moving suggests that, unless Latin America adopts active and sustained strategies to carry out the necessary social, economic, and technological structural changes, the mentioned technical potential is likely to materialize only in the most advanced countries, with the region running the serious danger of concentrating the perverse effects of the techno-economic revolution.

A prospective analysis was carried out, based on simple simulation models of the ecosystemic transformations associated to land use in each of the 18 major lifezones represented in Latin America. Two basic socioeconomic scenarios were defined by the whole region: the reference scenario and the sustainable scenario. The reference scenario suggests the type of environmental consequences associated with land use that an unrestricted and unregulated opening of the economies (in the context of an absence or widespread weakness of environmental and social policies) would have. The sustainable scenario shows that, from the ecological and technological points of view, it is possible to change direction toward a much more desirable long-term situation, without too large direct economic costs. Implications of strategic importance for the sustainable development of the region are identified.

Introduction

At the turn of the millennium Latin America is breaking away from its recent past. The pattern of import substitution is giving way to a growing openness toward the global economy, and there is an increasing wave of privatization and market orientation within the national economies. This economic change is taking place in the context of a continental wave of political democratization, an apparent economic recovery from the "lost decade" (the 1980s), a growing social polarization, a worsening of environmental problems, and the most intense urbanization process on the planet.

The similarity in the patterns adopted by the countries of the region (except Cuba, where economic changes are nevertheless also happening) has been attributed to a growing realization of the merits of the market economy. However, the changes have been attributed by others to strong international pressures from the international organizations, the Bretton Woods institutions created by the industrial nations after the Second World War.

From either perspective, there is an agreement that the process of economic globalization within which Latin America is evolving is one of the most dramatic developments of our times. Between 1965 and 1990 world trade of merchandises tripled, and the global trade of services increased more than 14 times. Financial flows reached gigantic levels. More than a trillion dollars circulate in the world economy every day. This capital flow offers unprecedented opportunities for profit, as well as for loss. At the same time, however, global financial markets leave even the strongest countries with somewhat limited autonomy over interest rates, exchange rates, and other financial policies (UNDP, 1996: 8).

The aim of this paper is not to discuss the advantages or disadvantages of the prevailing economic situation, but only to analyze some of the possible environmental implications derived from the way of insertion of the countries of the region in the global economy. Latin America has some of the most precious remaining environmental resources on the planet and is following an accelerating process of resource exploitation (Sunkel & Gligo, 1980; Gallopín et al., 1991), which appears to intensify with globalization.

From the social viewpoint, the indicators of poverty and unemployment in Latin America, which had been decreasing since the 1950s, have been increasing from the 1980s (PNUD, 1989; ECLAC, 1995). Although during the five-year period 1990–1994 some countries registered progress in their fight against poverty, others did not. In addition, starting from 1994, some worrisome trends appeared in some of the countries mentioned as successful above. Also, the rhythm and characteristics of the current economic growth continue generating less employment than necessary to productively absorb the growing work force (ECLAC, 1995).

The incidence of poverty in the region increased 5% in the short period 1985–1990 (a period of "economic recovery"—UNDP, 1996: 60). More recently, ECLAC (2000) estimated that the number of households in poverty decreased from 41% to 36% between 1990 and 1997, but the absolute number of poor increased slightly. The World Bank (2000), using additional data, reported that between 1987 and 1998 poverty, in relative terms, remained roughly constant, but the number of poor rose by about 20% (or 14.5 million).

In only 12 countries of Latin America and the Caribbean was the per capita income in the 1990s higher than that obtained in the past; in 22 countries of the region, the current income levels were reached in previous decades, which suggests an economic decline or stagnation. What is more serious is that the latter include more than 85% of the regional populations (UNDP, 1996: 3). The growth of the region in real per capita income was 2.9% per year in the decade of the 1960s, 3.7% in the 1970s, –0.7% in the 1980s, and 1.0% in 1990–1993 (UNDP, 1996: 14).

Income distribution improved in Colombia, Costa Rica, and Uruguay, and it worsened in Argentina, Bolivia, Brazil, Peru, and Venezuela (UNDP, 1996: 17). Income inequality also increased in Mexico, a country that quickly liberalized its economy beginning in the mid-1980s. In 1984, before the reforms, its Gini coefficient (an indicator of disparity) was 0.43, but by 1992 it had increased to 0.48. And in Chile, one of the most open economies in Latin America (and considered to be an example of success by many international financial organisms), income inequality has been increasing markedly from the 1970s. In 1970 its Gini coefficient was 0.45, but by 1990 it had increased to 0.57 (Berry, 1995). These negative social impacts have occurred while the region adopted increasingly market-driven economies, which has led many to believe that market economies can exacerbate economic inequality and poverty in the region.

It should be pointed out, however, that the increase in economic disparity is not unique to the Latin American region, but can be considered a global phenomenon (WRI, 1996). Income disparity appears to have increased in many countries that opened their economies, while the market has become a dominant economic institution globally. For example, in the last three decades, the share of global income by the poorest 20% of the population of the Earth decreased from 2.3 to 1.4%, while that of richest 20% increased from 70% to 85%. That doubled the ratio of the shares of the richest and the poorest—from 30:1 to 61:1 (UNDP, 1996). The differences among the developed and developing countries are moving, according to the latter report, " from the unjust to the inhuman."

The growth in disparity appears to be associated, at least partially, to the globalization of the economy. The patterns differ by region. Several countries of East Asia became successful examples of export-driven development, combining fast economic growth with low inequality and a high level of human development. In contrast, many sub-Saharan African countries have been increasingly marginalized by the forces of globalization.

A similar phenomenon is observed within countries. For example, by the late 1970s China began to liberalize the markets, privatizing the economy and opening rapidly to trade and international capitals. In 1979 its Gini coefficient was 0.33 (smaller than in any other country of East Asia). By 1988, it had gone up to 0.38—surpassing that of Indonesia and the Republic of Korea. And inequality keeps increasing, especially in the coastal area, which is the most directly bound to the world economy (Tabatabai, 1995).

From an environmental point of view, the deterioration in the Latin American region seems to be increasing (Gallopín et al., 1991; Gallopín, 1995; PNUMA et al., 1990; CDMA-ALC, 1990; UNEP, 1999). Although there has been some recent progress (particularly the elimination of economic incentives for deforestation in Brazil), the general situation continues worsening.

The question addressed here is: How will the evolution of the globalization process affect these tendencies of Latin America? Do opportunities exist, within the new context, to achieve sustainable development? We will argue that there are

opportunities to turn the tide of environmental degradation. Simplifying for the purpose of this discussion what is a rather complex issue, this paper will analyze two contrasting scenarios, one that leads to positive economic, environmental, and human outcomes, supported by "the Knowledge Revolution,"[1] and another that leads to lower performance on all these counts.

While studying the changes in the Latin American region, it seems worth taking into consideration that these changes take place in an international context that is far from stable. There are reliable indications that the world is moving through a widespread period of turbulence and change that could lead to a breakdown of historical trends and the emergence of new possible futures for the global system. Some of these futures are alarming, but others represent positive opportunities. The idea has been advanced that the world economy is moving toward structural changes that are so deep that the only appropriate metaphor is the concept of "punctuated" evolution, which some call a "Knowledge Revolution" (Thurow, 1996; Chichilnisky, 1996, 1998).

Among the processes propelling global change are:

- The collapse of the Soviet system, the end of the Cold War, and the almost universal expansion of market-oriented economies, also affecting countries that remained Socialist such as China, some other Asian countries, and Cuba.
- An unprecedented demography, with a fast-growing juvenile population in poor countries (according to the United Nations, 97% of the population increase between 1994 and 2015 will occur in the developing countries; United Nations, 1994), and an aged population economically dependent on the social security system in rich countries. The emergence of the "global teenager" (Schwartz, 1991) represents an enormous potential force of change, amounting to about 2 billion members by the year 2000 (in a world increasingly interconnected) and whose behavior is unpredictable.
- The techno-economic revolution supported by knowledge-intensive technologies, a revolution that is transforming not only the production process but also the social structure, as well as generating a global information economy and an unprecedented global connectedness (Chichilnisky, 1996, 1998; Herrera, 1986).
- The growing environmental degradation and the emergence of truly global environmental problems (such as climatic change).
- The growing social polarization between and within countries.
- The globalization and transnationalization of the economy, with growing influence of the big corporations, the creation of new commercial blocks, and the relative weakening of the nation-state.

[1] See, e.g., Chichilnisky (1996, 1998), who has been awarded a trademark for the use of "the Knowledge Revolution."

Two Global Scenarios

Globalization is a phenomenon that exceeds the strictly economic aspects to include environmental, technological, political, and cultural dimensions. It can result in radical transformations of the global system in the next decades. One possible trajectory represents scenarios of "barbarization" characterized either by a widespread societal decomposition and fragmentation, associated with high political and economic turbulence, or by an authoritarian world where the rich minority keeps (or attempts to keep) the rest of the population under conditions of low consumption (Gallopín, 1990a, Gallopín & Raskin, 1998).

In the other extreme there is a positive scenario (a Great Transition) where the emergence of a new sense of global solidarity, combined with the deployment of the potential of the new technologies and the empowerment of the civil society, leads to a new planetary order and qualitatively improved development paths.

While it is not possible to predict the future, it seems safe to predict that the "business as usual" scenario, based on the continuation of the historical tendencies and with indefinite economic *material* growth, is the less likely to occur. It appears to be intrinsically unsustainable in environmental terms. The scenario for Latin America that actually develops, as well as the way the region inserts in the global economy, will be very different depending on which global scenario materializes.

Historical Background in Latin America[2]

At the beginning of the century the Latin American economic system was entirely based on the production and export of primary products, and hence highly vulnerable to changes in the world economy. The crisis of 1929, prolonged by the Great Depression and followed by World War II, led to serious export constraints, forcing the countries to redefine their development patterns. This can be considered an immediate precedent to the situation of the region at the end of the 20th century, in which about 70% of the region's exports are still resources.

All countries (with the exception of Argentina) were forced at some point to suspend the service of the foreign debt.

Significant environmental changes took place between 1950 and 1980. The most important were (1) the expansion of the agricultural area allocated to short-cycled crops, (2) increase of the areas of permanent pastures, and (3) reduction of the areas covered by forests and expansion of the urbanized areas. These environmental changes were due to the widespread adoption of a new development model by the states of the region and the emphasis given to urban-industrial development and private foreign investment. This was driven by economic growth based on the expansion of internal markets, and was obtained through industrialization policies focused on import substitution using strong protectionist measures, overvaluation

[2] This brief historical synthesis is based partially on PNUMA-AECI-MOPU (1990).

of the national currencies, and facilitating imports of capital goods to develop the industrial infrastructure. This process generated a remarkable economic bonanza, also reinforced by a general improvement of the terms of trade of primary products.

The new productive strategies had a high environmental cost; the environment was subordinated to the need to accelerate growth. In this period the large cities grew quickly, as a consequence of the employment generated by the industrial sector and the expulsion of labor from the rural areas where the new agricultural technologies, added to problems of land tenure, displaced the traditional worker. Marginal areas grew around the cities, and an informal urban labor sector arose.

Also during this period the agricultural frontier unfolded, fundamentally at the expense of the forests. In the tropical areas cattle raising expanded in order to respond to a sustained external demand and to the internal demand of the high-income urban sectors. Cattle raising generated deep environmental degradation due to the use of land unsuitable for that activity and to the colonization of forests with inadequate technologies that led to their fast deterioration. In addition to the environmental costs there were high social costs, associated with agricultural modernization: displacement of the traditional productive systems reducing rural employment, expulsion of the population, and social polarization.

What has been described so far is representative of the model of import substitution. The economy felt the effects of the first fluctuations of the balance of payments, the public deficit, and the economic perturbations in the early 1970s. During this period the development strategy generated a modern, consumerist, and export-driven social sector and an increasingly marginal low-income sector.

Internally there were strong inflationary processes and indebtedness due to the use of external credit to develop the industrial and financial sectors, low redistribution, and authoritarian non-elected regimes that expelled many people out of the system contributing to the impoverishment and marginalization of important sectors of the population.

External constraints were associated with the restriction of markets in the industrial countries leading to a retraction of world trade, a deterioration of the terms of trade, the unfavorable insertion of the region in the international market, the global techno-economic revolution, and the increase of international interest rates with enormous impact on the foreign debt.

Following the first period, since the mid 1980s a new model spread in the Latin American region. The new model emphasized the liberalization of trade, non-traditional exports, and at least in some cases, foreign exchange and financial liberalization. Significant privatization took place in many sectors of the region. An effort to attract foreign capital was evident, as well as the expansion of the transnational companies. In many countries inflation decreased dramatically, and the rates of economic growth, depressed during the "lost decade" of the 1980s, showed important increases. As mentioned before, inequalities increased, and the population below the poverty line grew; there are indications that the economies of the region became more vulnerable. Several regional free trade agreements were cre-

ated, such as the North American Free Trade Agreement (NAFTA) among the U.S., Canada, and Mexico, and the Southern Common Market (Mercosur).

Latin American exports of merchandise grew at an annual average rate of 3% in the 1980s, and by the end of the decade the region attracted around a third of the private flows of capital to the developing countries (UNDP, 1996: 17).

The environmental effects of the new economic model followed since the mid 1980s are not still completely documented or even understood. As many ecological processes unfold through several decades, there may be impacts that have still not been perceived. Some of the clearest examples of the environmental impacts of this model are agricultural intensification of cash crops for export, in many cases in unsustainable form (as in the Brazilian cerrado; da Silva, 1994), the externalization of environmental costs that makes it possible for the soybean produced in the Bolivian agricultural frontier to be exported to Colombia at lower prices than the Colombian soybean (in spite of an exceedingly longer transport route) and the dismantling of the infrastructure (wire fences, water tanks) associated with the rotation between cattle and crops in the Argentinean pampas, thus shifting from a relatively sustainable production mode to one that shows growing problems of environmental degradation.

Environmental Issues in Latin America

The region as a whole is very well endowed in terms of natural resources. With little more than 8% of the global population, Latin America has 23% of the potentially arable land, 10% of the cultivated land, 17% of the pastures, 22% of the forests (and 52% of the tropical forests), and 31% of the permanently usable freshwater. It has not less than 3% of the world reserves of fossil fuel and 19% of the technically usable hydroelectric power (Gallopín et al., 1991).

This favorable regional profile masks, however, important internal differences. Demographic pressure is high in some of the countries, and low in others. There are countries in which the current cropland and even the potential arable land is scarce in comparison with the present population and that projected for the future. Latin America as a whole could only feed about 40% of the population forecasted for the year 2030, by using a low level of inputs. More than half of the South American countries (Bolivia, Chile, Colombia, Ecuador, Paraguay, Peru, Suriname, and Venezuela) and practically all Central American countries should surpass the intermediate level of inputs (equivalent to the one used currently in the region) and 11 countries (Ecuador, Peru, Suriname, Venezuela, the Bahamas, El Salvador, Guatemala, Haiti, Jamaica, Santa Lucia, and Trinidad/Tobago) would not feed their population of 2030 even using a high level of inputs (recalculated from the basic data of the model FAO-FNUAP-IIASA, 1984, by Gómez & Gallopín, 1995).

Many and serious environmental problems exist in Latin America and the Caribbean, as well as a number of not yet used opportunities.

All analyses of the recent history of Latin America indicate very high and fast rates of ecological deterioration, which appear in the form of deforestation, deser-

tification, erosion, and loss of fertility of soils; agricultural, industrial, and domestic pollution; accumulation of wastes; and growing vulnerability to landslides, droughts, and catastrophic floods (Sunkel & Gligo, 1980; Dourojeanni, 1982; Gallopín, 1995). The problem does not consist of the mere transformation or alteration of the natural ecosystems, but in the modality and result of these transformations that imply a degradation of the ecological base of the production, a true impoverishment and destruction of the renewable natural resources and the vital ecological processes of the region. Many of these alterations, such as desertification and soil erosion, are irreversible in practical terms. On the other hand, the environmental problems in the human settlements are very serious, and they are worsening.

Efforts have been made to identify and prioritize the main regional environmental topics in Latin America (CDMA-AL, 1990; Gallopín et al., 1991). The two dominant topics at the present time are those associated with land use and those associated with the urban environment, in the face of which the other ones (although important in themselves) become secondary in relative terms.

Globalization and the Environment

The globalization process, as already indicated, exceeds international trade and its broader economic dimensions. This article focuses on the economic and technological aspects, in the belief that globalization is a phenomenon intimately associated with the techno-economic revolution.

There are differing views on the impact of international trade on the environment. Some economists believe that free trade has a beneficial influence, because they foresee negative consequences from protectionism; see, e.g., Repetto (1993). On the other hand, a study of the environmental implications of the changes in the export profile of nine countries of Latin America and the Caribbean for the period 1980–1995 suggests that the opening of the economies in these countries resulted in an increased participation of dirty and natural-resource-intensive sectors (Schaper, 1999). In this context, it is often argued that trade leads to increased economic growth, and that growth itself has a positive impact on the environment. This argument, which is developed further below, does not take into account the problem that certain types of economic growth generate, by themselves, environmental degradation, and the existence of irreversible environmental damages (as in the extinction of species) (Ekins et al., 1994; Røpke, 1994).

Those who propose market regulation argue that investment could move toward the developing countries that offered lenient laws on environmental use, with possibly negative consequences for the environment. However, using existing data, supporters of free trade have observed that international investment has not moved preferentially to those developing nations that become "pollution-heavens" (e.g., Grossman & Krueger, 1991; Lucas et al., 1992).

In general terms, therefore, the free competitive market need not by itself produce incentives for environmental destruction although, in specific circumstances,

it could. Leaving aside general principles on the positive and negative impact of markets on the environment, we will concentrate instead on the specific circumstances prevailing today, in order to elucidate the actual connection between trade and the environment.

We focus on the specific structure of trade between industrial and developing nations as it is today, and as it has developed over the last century and particularly since World War II. Called generically "North-South trade," this pattern consists mostly of resource-intensive products (Chichilnisky, 1994a, 1995, 1995–1996, 1997; Chichilnisky & Heal, 1998) exported by developing nations to industrial nations in exchange for capital-intensive products. A generally accepted view is that an expansion of such a resource-intensive pattern of international trade can have damaging effects on the global environment by intensifying the extraction, exports, and consumption of natural resources.

Recent results have helped elucidate how this pattern of North-South trade may have emerged and developed, and how it could be overcome. One commonly accepted view is that the pattern of trade we observe today could have emerged from historical circumstances in which developing nations, which are in great measure agricultural societies, treat natural resources as "common" property while industrial nations, which have completed the industrial revolution, treat resources more as private property (Chichilnisky, 1993, 1994). This work showed that when such differences exist in property rights between the two regions, it leads to developing nations specializing in resource-intensive exports that are sold to industrial nations at prices that are below replacement costs. Under these conditions, it was shown that an expansion of trade amplifies environmental problems in developing nations, as well as globally, through a global version of the "tragedy of the commons" (Chichilnisky, 1993, 1994). When developing nations are exposed to large international markets for inexpensive and abundant natural resources from the industrial nations, the extraction of their common property resources intensifies and deepens, leading to lower prices and global consumption that exceeds efficient resource use. A case in point is the oil market, consisting mostly of exports from the South to the North. In this market oil sells at relatively low prices leading to the overuse of petroleum worldwide, delaying the commercialization of alternative clean technologies and increasing problems with the emission of carbon dioxide into the atmosphere (Chichilnisky, 1994a, 1994b, 1995, 1995–1996, 1997, 1998; Chichilnisky & Heal, 1998).

Under the specific conditions that prevail today, therefore, the standard assumption that countries can obtain mutual benefits through specialization and trade according to their comparative advantage must be revisited. When resources are common property in the exporting nations, a generally accepted condition in developing nations today, the market prices of resources are artificially low giving an impression of relative advantage even where none exists (Chichilnisky, 1993, 1994). In such situations some of the fundamental premises for effective market functioning, namely the existence of private property rights in all traded goods,

are not fulfilled in the reality. As stated by Daly and Goodland (1994), if markets were perfect and capital were immobile internationally, then unregulated trade in products would be advantageous for all nations. But in the real situation prices do not generally reflect social and environmental costs[3], and the benefits from unregulated trade may not be achieved.

A question that arises is whether the process of growth and industrialization can by itself ameliorate this situation in developing nations. It is generally argued in this context that growth and economic liberalization are good for the environment because the preferences of the consumers and the structure of the economy change as a country develops so that development brings new (often cleaner) technologies and that growing economies can invest more easily in environmental improvements. According to this line of thought, the countries in early stages of development necessarily concentrate on basic production and improvements of the infrastructure, accepting the associated environmental cost. This argument is based on empirical correlations between environmental degradation and per capita income (also called "environmental Kuznets curves"), which suggest that economic growth worsens environmental conditions until a certain point, but that at higher income levels, additional economic growth is associated with an improvement of environmental conditions (WRI, 1996).

Although some such environmental indicators as access to drinking water, urban sanitary conditions, and urban air quality show an improvement with increasing income, other indicators show a progressive deterioration (for instance, the emissions of carbon dioxide and the per capita production of urban wastes). In reality, the data show that the Kuznets curve is valid for local pollutants, such as air particulates, which appear to improve as the countries achieve higher levels of income. However, for global environmental problems such as greenhouse gas emissions, ozone depletion, and biodiversity destruction, the connection between income and environmental quality is generally reversed. The industrial nations have by far the largest negative impact on the global environment, through biodiversity destruction, ozone depletion, and greenhouse gas emissions (Chichilnisky, 1994a, 1995, 1995–1996, 1997, 1998; Chichilnisky & Heal, 1998). In addition, a substantial body of theoretical and empirical work shows consistently that the income elasticity of demand for environmental assets is lower than one, which means that poorer people tend to spend a larger part of their income on environmental quality than do richer people (for a review see, e.g., B. Kriström, 1996). It seems, therefore, that the connection between income and the environment cannot be described in simple terms[4]. Even in those cases in which Kuznets curves are applicable to environ-

[3] Other relevant analysis of the underlying assumptions and the environmental and social implications appear in Runnalls and Cosbey (1992), and in the special issue of the journal *Ecological Economics* (Vol. 9, No. 1, 1994), dedicated to Trade and Environment.

[4] The environmental Kuznets curves, based on empirical correlations, do not take into account the possibility that environmental degradation can harm the possibilities of future economic growth, or the possibility that part of the reduction in pollution observed in the industrialized countries could be due to the transfer of polluting industries to developing countries, a process not replicable by the latter (WRI, 1996).

mental indicators, it would be expected that economic growth in those countries (and for those variables) will continue increasing pollution, because the great majority of the world population has mean incomes below the inflection points of the curves. On the other hand, it is difficult to anticipate the impact on these curves of economic polarization associated with globalization (in other words, what will happen when and if economic growth becomes negative for some groups and countries).

The general argument from the environmental point of view is not that international trade is negative and that autarchy is desirable, but rather that under a variety of situations prevailing in today's world economy, which includes a variety of market imperfections, some market regulation may be necessary to reach a "sustainable free trade" (De Bellevue et al., 1994) or a "balanced trade" (Daly & Goodland, 1994).

The technological aspect of globalization is often associated with a third Industrial Revolution. This is also called a Knowledge Revolution because it is led by microelectronics and information technologies, and accompanied by a constellation of developments based on new technologies that use science and human knowledge intensively, such as biotechnology, new materials, new energy sources, and nanotechnology. This new wave of innovation is unfolding at a vertiginous pace, and the socio-economic changes associated with the emergence of the economy of information is leading to drastic transformations of human societies, some of them difficult to imagine today.

The development and diffusion of the new technologies in the region have the potential to produce very significant environmental changes (both beneficial and detrimental, direct and indirect). It is possible to anticipate that such changes will generate important impacts on the Latin American ecosystems, implying major effects on the ecological sustainability of the productive activities, alterations in the subregional water and nutrient cycles, changes in agricultural yields, disappearance of some ecosystems and emergence of new ecosystems, changes in the ecological supply of natural resources, and modifications in the limiting factors and ecological constraints.

Direct ecological impacts will result from using new technologies in food, industrial, and energy crops; exploiting new natural renewable and non-renewable resources; the creation and dispersal of new biological forms; and the emission of new substances into the environment. An attempt of identification of possible direct effects appears in Gallopín (1995a).

Indirect ecological impacts will result from social, economic, political, and demographic rearrangements associated with changes in price and demand, in the social organization of work, in the systems of production, in employment, in the international division of labor, in services, and in the relocation of human settlements and industries. Indirect ecological impact induced by the diffusion of the new technological wave is likely to be more extensive and pervasive than direct impact.

From the point of view of their environmental implications, many of the new and emergent technologies exhibit interesting differences with the previous para-

digm. The strategically important attributes of the new paradigm can be characterized as: ambivalence, flexibility, and knowledge-intensivity (Gallopín, 1995a).

Ambivalence. It is clear that information applications, microelectronics, and telecommunications can be used to centralize both information and the power of decision and control, but they also have the potential for decentralization of the decisions, to increase participation, and to link isolated and remote areas; biotechnology can favor the monopolic concentration of large-scale agricultural production or it can be applied to increase the yields of the subsistence farmers of small scale.

Flexibility. The new technologies allow (potentially) easier adaptation and adjustment to the local social and ecological conditions (an extremely important aspect for the sustainability of development) than the previous technologies. The information technologies make possible (and economically efficient) the implementation of new operational modes, such as "flexible manufacture," "tailored" or "on-demand" production, and the minimization of stocks ("zero inventory"); these modes are incorporated in the recent concepts of re-engineering of organizations. This leads to the scale of the plant becoming increasingly independent of the scale of each market, and productivity increasingly independent of the scale of the plant, with deep changes in the defining factors of competitiveness (Pérez, 1986). This last element implies a very significant difference with the previous paradigm: in many cases the new technologies are not (inherently) associated with economies of scale.

Knowledge-intensivity. The new technologies are generally (in fact or potentially) much more efficient in the use of energy and materials than the modern technologies originated in the postwar period. They can be can qualified as "knowledge-intensive" or "science-intensive" technologies, rather than capital-intensive, energy-intensive, or material-intensive. The new technologies are reducing the ratio of raw materials/product, are substituting materials (a clear example is the substitution of the much more efficient optic fiber for copper wire in communications; interestingly, the material base of microelectronics is silicon, one of the most abundant elements on the planet); and they are increasing the efficiency of other processes (the case of electronic regulation of fuel combustion in automobiles). All of this can contribute to the currently visible process of relative dematerialization of the economy, to the conservation of natural resources, and to the reduction of pollution per unit of production or consumption.

Both economic globalization (in the sense of the elimination of trade barriers among countries) and the new technologies possibly represent elements of an unavoidable stage in the evolution of civilization. These processes obviously have the potential of greatly improving the living conditions of the population, to support the rational use of planetary ecological and human resources, and to reduce military conflicts. However, the distance from potential to realization is a long one. In an asymmetric world where inequalities continue growing, there is no guarantee that the potential benefits will be obtained by all or by the majority of the population. It is often argued that some social cost is unavoidable in any historical transition, and that transitory sacrifices will be compensated by a general improvement. Again, this may be so, but it is not guaranteed. In an asymmetric

world, the risk of a monumental *unnecessary* cost in human suffering of the techno-economic transition is high, and the risk of a consolidation and accentuation of the inequalities driving to a "global barbarization," as described in Gallopín (1990a) and Gallopín & Raskin (1998), is not negligible.

Although globalization has often helped growth in the strong countries, it has bypassed the weak ones. The share of world trade for the poorest countries, including 20% of the world population, has fallen between 1960 and 1990 from 4% to less than 1%, and they receive a mere 0.2% of the world's commercial lending (UNDP, 1996: 9).

The total flows of capital to the developing countries tripled between 1987 and 1994. Their composition moved markedly from official development assistance (ODA) and toward private capital flows, which increased from a participation of 37% of total flows to 76%. In real terms, ODA fell 9% between 1985 and 1993. The seven-fold expansion of private flows to developing countries, from 25.1 billion in 1987 to 172.9 billion in 1994, could be seen as compensating to some degree the fall in ODA. But private funds have generally ignored the most capital-scarce developing countries, going instead toward the semi-industrialized "emergent markets." Of the total flows in 1993, 68% went to Argentina, China, Mexico, Singapore, and Turkey. Direct foreign investment (which not only provides fresh capital but also contributes higher technological levels) is also concentrated: it is estimated that a record 37% (84 billion dollars) arrived in developing countries in 1994. Nearly 40% of this went to China. Another 24% went to Hong Kong, Indonesia, Malaysia, Singapore, and Thailand. By contrast, sub-Saharan Africa only received 3.6%, and the least developed countries only 1% (UNDP, 1996: 78).

From a historical perspective, it is important to recognize that both globalization and the Knowledge Revolution do not represent (by their origin) a transition to a new social formation but a revolution taking place within, and generated by, the industrial capitalist society. This current process is associated with the emergence of a new economic, social, and cultural pattern, which can be interpreted as the response to the inability of the paradigm of resource-intensive industrialization that emerged in the postwar period to insure the continued and sustainable economic growth in the industrialized countries[5].

Certain tendencies have been observed, including the increasingly asymmetric income distribution between social groups and nations, and the loss of some national autonomy of those countries in which the largest organizations (specifically

[5] Nochteff (1987) provided a lucid analysis of the origin, trends, and proven as well as likely socioeconomic impacts of the current techno-economic revolution on Latin America. His analysis concentrated on the "electronic complex» (microelectronics, information sciences, telecommunications), which constitutes the nucleus of that revolution; however, the main conclusions can be extended to other new technologies. Those share several important characteristics and interact synergically with each other. Besides, biotechnology, new materials, and the new energy sources would tend to subordinate to the technological system centered in microelectronics (Pérez, 1986). In general terms, the main direct impact of microelectronics will concentrate on services and manufacturing, while biotechnology would affect most directly agriculture, mining, and the primary sector in general, and the chemical industry. In this sense, the development of biotechnology fills a gap left by the complex of information technologies; both technologies are complementary at several levels.

the transnational companies and the most powerful states) do not have their main nucleus of decision-making, operations, and development (Nochteff, 1987). To this we may add the opening of the services sector to international competition, which leads to a centralization of the power of the transnational corporations, which already control about 70% of world trade (Daly & Goodland, 1994). These tendencies are in general negative for Latin America and are likely to affect the region if the current (exogenously determined) technological change continues.

The current situation of Latin America vis-à-vis the globalization process and the new technological revolution is very different from that of the industrialized countries. At the same time that a minority of the world population lives in the "post-industrial" civilization, in Latin America and the Caribbean three different technological waves coexist: the new technological wave, the industrial revolution, and even the agricultural revolution (large numbers of peasants survive at pre-industrial technological and production levels). As indicated by the United Nations Development Program, one of the risks of globalization is that the groups and people least able to adapt to the changing market conditions with its new technologies and skill requirements will be further marginalized (UNDP, 1996: 103). This is directly applicable to the growing marginal population of the region.

The global environmental issues will surely represent in the future one of the main factors of interdependence between the North and the South, and by the same token, a space for negotiation[6] between the industrialized and the developing countries. By contrast, in many other aspects the techno-economic revolution facilitates an ever-increasing autonomy of the advanced countries with regard to the world in development and the planetary level.

It is clear that any loss of autonomy by the Latin American countries in defining their production, consumption and distribution patterns, and increasing concentration of power in transnational corporations, would bring with it an additional weakening of the feedback between economic activities and ecological deterioration[7]. This could accentuate a tendency toward the overexploitation of some natural resources, the sub-utilization of others, and the externalization of ecological costs from the large organizations toward the region.

A scenario that raises concerns is that the new techno-economic paradigm as is now unfolding could lead to certain disequilibrium between structures of production, gearing even more production toward exports and toward the demand of minoritary high-income sectors, with pressures to generate new demands and re-

[6] For example, one of the negotiation arguments (Gallopín, 1990b) could be directed to the international opening of the access to the new technological developments, to favor the change of the processes of industrialization of the developing countries, as a prerequisite to reduce the global ecological impact associated with growth. The argument has logical validity, if it is considered that today's developed countries are fundamentally those that consumed the planetary ecological potential, and that the current degradation of that capital will restrict the possibility of growth for the developing countries (if they are constrained to use the traditional paths of "dirty industrialization"). The importance of wide availability of new technologies is recognized in Agenda 21 (Chapter 34), but little has been accomplished.

[7] This feedback is essential for the stability of the interactions environment-development (Gallopín, 1980).

duce the useful life of durable goods, thus increasing the production of wastes[8] and the marginalization of large sectors of population.

The orientation of production toward non-essential consumption goods and the obvious explosive tendency toward the increase of supply and diversity of durable consumer goods contribute to generate an indefinitely growing pressure on the environment and scarce resources diverted to non-essential uses[9]. This is more so considering that the trends do not favor a transition toward collective modes of consumption of the goods and services that allow it, but rather they exacerbate individual consumption, multiplying the number of units necessary to satisfy the demand.

A current trend appears to be the decentralization of industrial production, accompanied by centralization in the control of knowledge generation[10]. In the case of microelectronics, the investments of transnational companies in the region would be those dedicated to the realization of progressively less remunerative tasks and with smaller technological importance, often adopting the form of "enclaves" without linkages to the rest of the local productive system. In the area of materials, the apparent directions of change are the geographical relocation of the production of traditional materials in search of comparative advantages in the cost of energy, or to take advantage of savings in transport costs and the flexibility granted by the proximity to the source. Another direction of change is the growing diversification of the plants in the developed countries in the area of the new materials, which are more sophisticated and appropriable.

The main issue is to avoid the type of localization of industries and other productive activities that ignore the local ecological constraints and the environmental adequacy of the localization of activities, with the consequent worsening of environmental problems. To avoid a trend toward the relocation of industries with high polluting potential toward the developing countries, norms of environmental protection should be included in the treaties of international trade. It is of course possible that some ecosystems of the region could be used by big organizations as space for testing new technological developments with high environmental risk, or to explore comparative advantages of the germplasm or of the local ecosystems[11].

---

[8] This refers to wastes associated with consumption. As for production wastes, they could possibly diminish due to improvements in input efficiency facilitated by the new technologies.

[9] On the contrary, the production of basic goods has a natural ceiling determined by the satisfaction of the population's fundamental material needs.

[10] What led Celso Furtado to anticipate the possibility of a tele-guided destiny for Latin America (Furtado, 1984).

[11] This is not mere speculation. High technology has already been added to the list of well-known cases of pharmaceutical experimentation using the human population of the region (experimental birth-control methods, new drugs, etc.). The Wistar Institute of the U.S., with funding provided by private organizations (Laboratory Rhone-Merieux, Laboratory Transgene, Rockefeller Foundation), clandestinely carried out an experiment in 1986 in emplacements of the Pan American Health Organization in the locality of Azul, Province of Buenos Aires, Argentina. The experiment consisted of the inoculation to cattle of a new genetic recombinant rabies vaccine, obtained by genetic engineering. This was the first test in the world of that vaccine under field conditions. The experiment was hidden from the Argentinean government and its sanitary authorities, and, according to public accusations, both the workers who manipulated the inoculated cows and consumed their milk without pasteurizing, and the population of Azul (who consumed it pasteurized and marketed) were not informed. The experiment was interrupted by the Argentinean sanitary authorities when its existence filtered openly. The case caused a scandal in the U.S. (symptomatically, the public commotion was much weaker in Argentina). See *Revista Humor* No. 186, 187, 190 and 191, years 1986/1987, Buenos Aires.

It seems clear that the technologies and production patterns generated by the large organizations of the industrial countries cannot be expected to spontaneously adapt to the needs and potentialities of the countries of the Latin American region. This implies that the new imported technologies could be poorly adapted to the ecological cycles of the local ecosystems.

Another macro effect that can be expected from the exogenous determination of technology in Latin America is the potential for a lack of "sensitivity" of the structure of production toward the endowment of natural resources in the countries of the region, generating tendencies to the application of excessive pressures on some resources, and simultaneously to the neglect or sub-utilization of others. The rationality of the big transnational companies, as well as their capacity to mobilize capital in the planetary space, could induce levels of renewable natural resources use exceeding ecological regeneration rates, leading to the degradation of the productive ecosystems and their abandonment when their profitability becomes inferior to that of alternative places on the planet. The pressure to export and to compete internationally can result in strong stimuli to produce in inappropriate lands, as well as generate an internal competition with the land dedicated to the production of basic food crops.

The fundamental input for the new techno-economic paradigm is science, increasingly integrated into technology, the type of science that is more directly linked to the requirements of industrial societies. The WTO considers knowledge as private property instead of as the patrimony of humanity, and it favors the patenting of germplasm and living organisms. All these factors contribute to decrease the adaptability of the technologies to local potentials and ecological restrictions (except for those that are specifically designed to be programmed and adapted).

The widening of the income gap between the advanced countries and those of the region, and the structural tendency to the imbalance of the regional external sector, suggests a possible relaxation of the norms of environmental and ecological protection and an accentuation of the current tendency to the overexploitation of the productive ecological base for purposes of export. Those trends, combined with social tensions due to growing technological unemployment and regressive income distribution, are inducing so-called "war economies," abandoning the environmental (and social) objectives of development. On the other hand, it is possible that some regional trade agreements would stimulate a relative homogenization of environmental and social protection criteria. The risk is, however, that the homogenization of standard be made "downward" rather than "upward," restraining the growth potential of the region. As mentioned before, the combination of international mobility of capital and free trade of products stimulates an international standards-lowering competition (instead of increases in efficiency) in the name of reducing costs in order to attract capital (Daly & Goodland, 1994).

Other sources of concern are the tendencies to technological unemployment, which could lead to increasing marginilization and might even end up reverting, in some countries, the current net migration from the rural to the urban areas, with ecological consequences on the rural environment. In addition, the tendencies for

the polarization of income within countries could favor the increase of ecological deterioration associated with poverty, as well as that associated with over-consumption. Finally, the increase of the "vertical" connectivity between the global and the local associated with globalization could have unpredictable consequences. It appears that in many cases the growing integration of the world population to the markets increases dependence from factors increasingly distant from local control (Gallopín, 1994).

Currently, small peasant production is sensitive to distant factors operating through subtle channels. An increase in the U.S. interest rates can trigger policy changes across and along the developing world (because the costs of serving the external debt are linked to international interest rates). Many of those policies directly affect the life of the small producers, particularly the subsistence peasants. Their changing activities will also have environmental consequences and thus become part of the environmental impacts of the operation of the international financial system. Maletta (1988) analyzed several causal chains operating at different scales in the case of peasant agriculture in the Andean region.

From the environmental point of view, the consequences of the insertion of peasant producers into the market may vary drastically depending on the "economic rationality" of the producer[12].

There is another phenomenon associated with globalization and technology that has very deep systemic implications: in hierarchical systems (those containing components and processes that operate at different levels of aggregation and space and time scales) the lower-level subsystems operate faster (at a smaller time scale) than the subsystems and processes belonging to the macro level. Globalization is leading, on one hand, to the local systems becoming more and more connected to the global system; and on the other hand, due to the operation of a global network of telecommunications in combination with new concentrated decision-making systems operating at the planetary scale (particularly transnational companies), the dynamics of the global level is becoming faster, in several dimensions, than the dynamics of the lower levels (Gallopín, 1991). The consequences of this unusual phenomenon are theoretically rich, but largely unpredictable.

It is also difficult to anticipate the environmental impacts associated with the general redefinition of comparative advantages because of the possibility of emergence of new unsuspected advantages, and the probable multiplication (and increased volatility) of the number of factors that define comparative advantages.

[12] In a case-study in the Argentinean province of Chaco (Gallopín & Barrera, 1980) it was possible to verify the coexistence of two economic rationalities in the same geographical space. Most of the producers tried to maximize the rate of profit (capitalist rationality); the subsistence producers, however, sought to maintain constant total family income (rural rationality), a phenomenon originally detected by A. V. Chayanov in Russia (Chayanov, 1925). As a consequence, the rural capitalist producers intensified their exploitation (increasing the environmental impact) as a response to an increase in the price of their products, and diminished their activities in the opposite case. The subsistence producers reacted in inverse form; they increased the exploitation of the land (and their self-exploitation) when the prices went down, in an attempt to maintain their total income. Thus, both types of producers had opposite behaviors, both perfectly rational in the context of each objective function.

The reduction in the relative weight of wages in the new techno-economic paradigm could reduce the importance of the comparative advantages of cheap labor, affecting the prospects of the countries that based their growth on that factor. In the short term, however, globalization may result in a North-South labor competition, stimulating a tendency toward labor-intensive products and processes (Daly & Goodland, 1994). The reduction in the ratio of raw materials to product, and substitution of materials, would affect more directly the countries that based their process of capital accumulation on their mineral or forest resources. The new technologies (and particularly biotechnology) are already affecting the traditional agricultural producers (in the North as well as in the South), transferring profits and the control of production and commercialization to the big chemical and pharmaceutical transnational companies and the big trading companies. Increases in agricultural yields in the advanced countries (facilitated by the new technological developments) are reducing the soil and climatic comparative advantages, closing traditional markets for agricultural products of Latin America, and increasing international competition for those products from the industrial countries.

New Comparative Advantages?

Several comparative advantages could arise in the countries of the region, with different environmental consequences. The range includes the advantages related to the access to sources of cheap energy, those associated with the reduction of costs of transport because of the proximity to the sources of natural resources, those of localization granted by permissive environmental or sanitary legislation (a somewhat perverse advantage), and those of exploitation of the local ecological or climatic conditions or components, among others.

In ecological terms, this changing mosaic of comparative advantages in the countries of the region could lead to a major increase in the pressure of exploitation upon fragile or remote ecosystems currently untouched, the abrupt valorization of particular ecological elements or functions, and the devaluation of others, the installation of new biological forms and even ecosystems in the region, etc. In absence of social regulation, these phenomena can result in the overexploitation and degradation of the regional ecosystems, and in the loss of the traditional comparative advantages that could be associated with them.

The overall ecological prospects that can be inferred in this regional scenario are somewhat discouraging, although there exist some isolated positive aspects. Yet somewhat paradoxically, the *technical* potential for a sustainable management of the ecosystems, for the control, monitoring, and minimization of environmental pollution, for the adaptability of the plants and technologies to the local social and ecological conditions, for a spectacular increase in the production of satisfiers of human needs, for the diversification of uses of ecological resources, and for ecologically sustainable development, is probably higher today than in any moment of the past.

However, unless Latin America adopts active and sustained strategies—which are endogenously defined, and shared among social actors and countries—to carry out the necessary social, economic, and technological structural changes, the mentioned technical potential is likely to materialize only in the most advanced countries. Otherwise Latin America faces the danger of concentrating the perverse effects of the techno-economic revolution[13].

Although the analysis presented in this section concentrates on the possible regional ecological effects of the high technologies, the impact of the diffusion of technologies already existing (modern) and of the change of products should not be underestimated. Both phenomena are directly linked to economic globalization. The recent history of Latin America shows impressive shifts of products and technologies in the agricultural sector. This indicates that the ecological effects of the new technologies in Latin America will not replace those of modern and "traditional" technologies, but rather they will be added to them, at least during the next decades.

Analysis of Alternative Scenarios

The multiplicity of interlinked factors makes it impossible to provide a detailed prediction of the future environmental impact of globalization on Latin America. It is, however, feasible to explore future trajectories representing alternatives that seem plausible on the basis of the information now available.

In this spirit, a prospective analysis was carried out, based on simple simulation models of the ecosystemic transformations associated with land use in each of the 18 major life-zones represented in Latin America. The time-horizon considered was 50 years, beginning in 1980 (Gallopín, 1992, 1995; Gallopín & Winograd, 1995).

Within each life-zone annual land transformations are simulated as simple functions of land use, the general properties of the life-zone, and the selected scenario. Within each life-zone the following categories are distinguished:

"Natural": undisturbed areas with primary vegetation, but also including areas that have been perturbed in the past and today have a vegetation similar to the original one; "Altered": modified by human activities (forestry, shifting agriculture, ranching, etc.) with coexistence of portions of the original ecosystem and secondary vegetation, and including fallow from shifting and peasant agriculture; "Agricultural": areas annually sown and harvested, including permanent and annual crops and non-traditional plantations (e.g., coca and marijuana); "Grazing": areas with natural or artificial pastures, currently used for ranching; "Plantations": areas re-

[13] Even the optimistic analysis of Pérez (1986), who explicitly emphasized the new opportunities, referred mainly to the possibility of opening new free spaces for medium and small companies, to the technical potential for the improvement of production, the possibilities for decentralization, for new degrees of freedom, the diversification and adaptability potential, etc. But all this in a space dominated by the giant companies. She, too, signaled the unavoidable necessity of new development strategies for the countries of the region.

forested for forest exploitation or watershed protection; "Wastelands": areas with severe human-accelerated erosion and desertification processes with irreversible changes in their structure and function (it excludes the natural deserts); and "Urban": urbanized areas (essentially cities).

Two basic socioeconomic scenarios were defined by the whole region: the reference scenario and the sustainable scenario.

The reference scenario implies the partial continuation of the economic stagnation of the 1980s, followed by a moderate increase of regional economic growth. The development patterns remain the same but with a growing influence of globalization and of transnational companies upon the economy. The new technologies enter in the region exogenously. There is no significant move toward the implementation of environmental and sustainability policies. The emphasis of agricultural production is on cash crops for export, and secondarily on crops for internal consumption. The reference scenario also assumes a gradual reduction in the advance of the agricultural frontier in tropical areas, and a general intensification of land use.

The sustainable scenario contains an emphasis on endogenous decision-making by the region[14], and it includes policies to improve income distribution, implementation of active scientific, technological, and environmental strategies including a move against resource-intensive and toward knowledge-intensive growth (Chichilnisky, 1994), and development of new systems of agricultural production. The main emphasis of the agricultural production is on crop diversification for internal consumption, and only in second place, on cash crops for export.

The general characteristics of the two scenarios appear in Table 1.

Table 1. General hypothesis in the scenarios for the simulation runs.

Variable	Reference scenario	Sustainable scenario
Demographic growth	2.2% in 1980 to 1.2% in 2030	2.2% in 1980 to 1.2% in 2030
Production	0.5%/year	the range of 0.5 to 1.2%/year
Average growth of per capita agricultural yield	1%/ year	Within the range of 1.5 to 2.0%/ year
Animal carrying capacity (animal units)	From 0.6 UA/Ha in 1980 to 1.2 UA/Ha in 2030	From 0.6 UA/Ha in 1980 to 1.5 UA/Ha in 2030
Annual harvested area	65% of the agricultural area in 1980; 75% in 2030	65% of the agricultural area in 1980; 85% in 2030
Land-use allocation	Emphasis on export crops; secondarily, in crop diversification for internal consumption and for export	Emphasis in crop diversification for internal consumption and export; secondarily, in export crops

Source: Gallopín & Winograd (1995).

[14] Sometimes called "self-reliance": this denotes a capacity to make autonomous decisions, and it does not necessarily imply autarchy.

Although the scenarios were not designed specifically to simulate the effects of economic globalization, it is clear that the reference scenario represents a more passive attitude of the region toward globalization, while the sustainable scenario implies a more active and dynamic attitude.

The comparison of the calculations under the two different scenarios allows us to illustrate the potential impact of globalization (together with other variables) under either a passive or an active attitude of the region.

Given the simplicity of the models in comparison with the complexity of the ecological and social processes that are unfolding in the region, and the inherent uncertainties in the available primary and secondary data, the results proposed should be taken as indicative and not as definitive numeric results or attempts of detailed prediction. Some of the results have been revised based on later data (Winograd, 1995a), but the changes do not change the overall findings.

The detailed results of the simulations for each life-zone per decade may be consulted in Gallopín (1995a), Gallopín and Winograd (1995), and Winograd (1995). Here only the general results for the whole region are presented (Fig. 1).

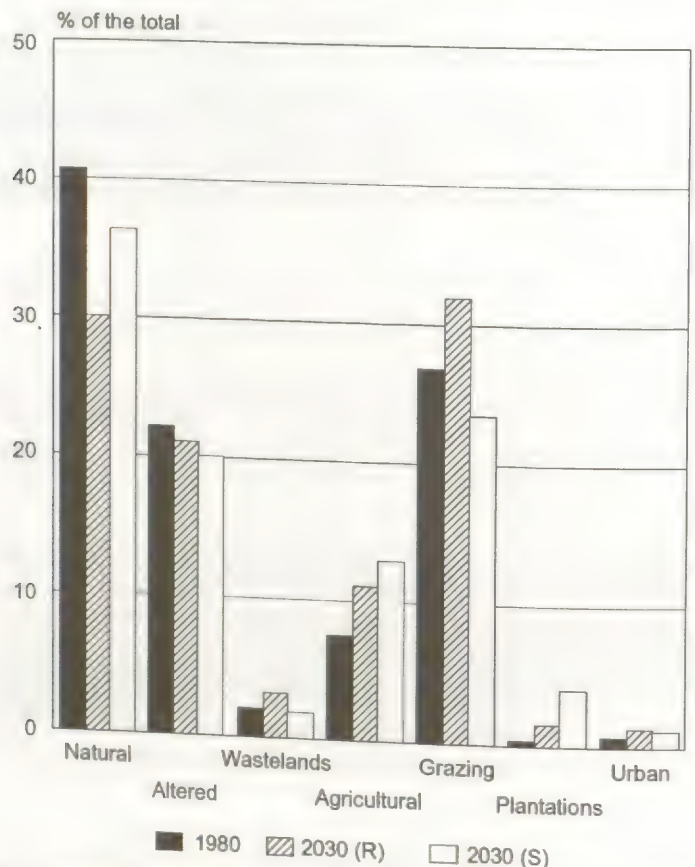


Figure 1. Surface under different categories in 1980 and 2030 under the reference scenario (R) and under the sustainable scenario (S). Source: Gallopín (1995a).

Reference ecological scenario

For the whole region the results imply the transformation of 5 million hectares per year (as an average for the 50 years' runs) of virgin and semi-virgin ecosystems. A fraction of 78% of this surface will come from the tropical areas, 19% from the subtropical areas, and only 3% from the temperate areas. As much as 45% of this transformed area will become agricultural (30% under shifting agriculture, 15% under permanent agriculture); 30% will be used for grazing, and 22% for forest exploitation.

Two major processes drive a large part of the dynamics: (1) the advance of the agricultural frontier, translating into a decrease of natural ecosystems and the growth of agricultural, grazing, and altered areas and (2) the intensification of land use which, in the dry zones, increases the wastelands at the expense of the altered ecosystems, and in the humid zones increases the area of altered ecosystems, within which subsistence agricultural activities intensify.

The total surface of altered ecosystems in the region diminishes because in many life-zones the stocks of land are being exhausted, leading to the intensification of land use instead of territorial expansion.

In this scenario the following environmental problems related to land use stand out:

Soil erosion, originated in deforestation, inappropriate agricultural techniques, overgrazing, and overexploitation. It will particularly affect the tropical and subtropical mountain rainforests and the subtropical rainforests of Central America, the Andean countries, and Brazil. To a lesser degree, the Argentinean pampas will continue to suffer from erosion.

Watershed degradation, due to deforestation and dam construction. It will affect mainly the tropical and subtropical mountain and lowland rainforests in Central America, the Andean countries, parts of South America, Brazil, and Mexico, as well as the temperate rainforests of Chile and Argentina.

Floods, due to watershed degradation, deforestation, and natural processes. They will mainly affect the tropical and subtropical mountain and lowland rainforests in Central America, the Andean countries, and Brazil, and some of the savannas, subtropical forests, and pampas of the Andean countries, Argentina, Brazil, and Bolivia.

Desertification, associated with overgrazing, excessive extraction of fuelwood, and cyclic droughts. It will advance mainly in the Patagonian steppes, the Puna, the dry tropical forests, the tropical and subtropical desert shrublands, and the temperate thorn scrublands in the Andean countries, Brazil, Argentina, Chile, Peru, Mexico, and Central America.

Agricultural pollution will continue in many of the cultivated lands in the whole region, and agricultural, industrial, and urban pollution will increase in the deltas and mangrove forests of Central America, the Caribbean, and parts of South America.

Firewood deficit will continue increasing in most of the ecosystems. The shortage of firewood due to deforestation and overexploitation of the forests will affect more than 50 million people in the arid areas and the Andean highlands in the next 30 years.

Sustainable ecological scenario

Under the endogenous scenario, the region is capable of satisfying the agricultural, livestock, fishing, and forestry internal requirements in a sustainable manner within the considered time-horizon of 50 years after 1980, with a substantial surplus for exports.

Three major processes account for a large part of the dynamics in this scenario: (1) Emphasis upon productive rehabilitation of deteriorated and altered ecosystems (which cover 22% of the total land area), because it represents the most realistic strategy for dealing with many of the complex tropical and subtropical ecosystems; (2) Impulse to integrated rural production systems (agriculture-animal husbandry-forestry-aquaculture) whenever they are appropriate; and (3) Active pursuit of integration of the new technologies into traditional and modern technologies.

Besides the quantitative differences with the pattern derived from the current trends, the qualitative changes in the modality of rural production imply a drastic reduction of the ecologically degrading processes previously defined.

For the whole region those figures imply the transformation of 2 million hectares per year of virgin and semi-virgin ecosystems (most of them in tropical areas). Protected areas represent 35% of the remaining natural ecosystems. Altered ecosystems will cover 20% of the area, the same figure as in the reference scenario. However, in this case most of the altered lands become productive (14% in sustainable forestry and 6% under rehabilitation). Cultivated lands increase to 13% (7% under intensive agriculture, 3% under agroforestry, and 3% under shifting cultivation). Grazing lands decrease because of increments in carrying capacity (18% is under intensive and semi-intensive grazing systems, and 7% is integrated with forestry). As a consequence of the rehabilitation and restoration activities, wastelands are reduced to half their initial surface.

An unplanned positive effect of this scenario is that, due to the strong emphasis on reforestation and agroforestry, by the year 2030 about 64 million hectares would have been reforested (mainly in the altered areas). This represents 14% of the world area that was estimated by Sedjo (1989) to compensate, if afforested, the excess atmospheric carbon generated by human activities. Note that the estimated current emissions of biotic origin (mostly deforestation) by the region represent between 8 and 10% of the world total (Gallopín & Winograd, 1995).

The ecological, technological, and economic feasibility of this scenario is argued in detail in Gallopín and Winograd (1992, 1995).

The reference scenario suggests the type of environmental consequences associated with land use that an unrestricted and unregulated opening of the economies in the context of an absence or widespread weakness of environmental (and

social) policies would have. This weakness characterizes the general situation of the region at the present time. The environmental problems would affect critically the economies and the societies in several countries of the region.

The sustainable scenario shows that, from the ecological and technological point of views, it is possible to change direction toward a much more desirable long-term situation, without too large direct economic costs[15].

The main unknown lies in the political feasibility of this scenario, since it goes against all the recent tendencies in the region and of the macroeconomic forces loosened in the last decades.

However, it should be taken into account that the current trends are clearly unsustainable, both ecologically and socially. According to some analyses, they may also be unsustainable in economic terms (The South Centre, 1996; Chichilnisky, 1996). It is therefore necessary to look for alternatives to the current trajectory.

Strategic Elements for Sustainable Development in Latin America and the Caribbean

From the analysis of the simulation models as well as of 21 real-life case studies, and a systematic evaluation of environmental opportunities and constraints at the regional level (Gallopín, 1995), the following elements of strategic importance for the sustainable development of Latin America and the Caribbean stand out:

- The analysis of current trends in the region clearly shows that the prevailing pattern of development is ecologically unsustainable and therefore not viable in the long term.
- There are no important ecological constraints (at the level of the region as a whole) for sustainable development, nor for the conservation of the areas required to maintain the essential ecological functions and services. However, in some countries there exist important ecological restrictions (such as shortage of arable lands), as well as clear environmental complementation, and therefore intra-regional cooperation will be essential.
- At the moment lack of technologies is not a critical obstacle for the sustainable development of the region (in the sense of representing a bottleneck at the regional level)[16]. This does not negate the need to fill the existing gaps of knowledge of appropriate management of some ecosystems.
- The new and emerging technologies can play a very important role for both environmental and general sustainability. Ecological analysis allows the identifica-

[15] The necessary direct investments related with land use would amount to less than 4 billion dollars per year for a 50-year period (Gallopín & Winograd, 1992). These annual costs represent 0.3% of the GDP of the year 1993 for Latin America and the Caribbean.

[16] Today there exist management techniques that are economically, socially, and ecologically sustainable for ecosystems as varied as the tropical humid forests, the tropical dry forests, the tropical montane forests, the Puna, the dry temperate shrublands, the temperate humid forests, Patagonia, etc. (Winograd, 1995; Gallopín & Winograd, 1995).

tion of broad regional priorities for research and development (R&D), taking into account the major ecological opportunities and constraints for development (Gallopín, 1995a). Many of them are absent from the current official priorities set by the scientific systems of the countries of the region.

- Latin America and the Caribbean are a region of high ecological, social, and productive heterogeneity. Different social actors and production types coexist in dissimilar environments. A strategy based on *technological pluralism* (complementary use of traditional, "modern", and high technologies) is essential for sustainable management of heterogeneity.
- *Productive pluralism* (the coexistence of different types of rural production systems, integrated through local, national, and regional policies) represents a more appropriate alternative than productive homogeneity from the point of view of sustainability of development.
- In terms of environmental sustainability the concept of *technological blending* (constructive integration of new and emerging technologies into traditional or modern technologies) assumes particular importance, requiring new forms of organization and an integral strategy for technological innovation and diffusion.
- The domain of application of the new sophisticated technologies is not confined to the "modern" (essentially urban-industrial) sector of the economy. These technologies can play a very important role, given the current context of the region, in the generation of new solutions to problems such as critical poverty, using science and high technology to develop simple, but new and effective technological solutions accessible to the poor, or in the reformulation and revaluation of native technologies used extensively in the region. This implies using high technology to develop solutions of "sophisticated simplicity."
- An important strategic principle is that of the integration among the different areas of the new technologies. It is often assumed that the field of application of biotechnology is limited to agriculture or the pharmaceutical industry, and that the field of application of information sciences is the services and industrial sectors. The integration among areas of new technologies can facilitate very important synergies (for example, in peasant agriculture; Gallopín, 1995a).
- Due to the multiple interlinkages among factors associated with globalization, from the speed and turbulence of changes to the unpredictability of many of them, the development of strategies directed to create a generic social and ecological capacity to respond to changes in a flexible, adaptive, and proactive form will be critical for the region. This general capacity may be equally or more important than specific measures implemented to confront particular challenges.
- An important strategic principle to cope with the growing uncertainties and the lack of sufficient resources to address all possible negative impacts of globalization is to try to develop "*tie-in strategies*" whenever possible. The concept, proposed by Schneider (1989) in the context of global climatic change, is also valuable with reference to the environmental impact of economic globalization. It implies giving preference to those actions directed to avoid or to adapt to

negative changes (or to benefit from positive changes) of economic globalization, that will still provide clear social and ecological benefits even in the case in which the anticipated changes do not materialize (or other unexpected phenomena occur[17]).

Conclusions

Latin America is changing in the context of major global trends. These include, on the negative side, an enormous foreign debt, an increasingly regressive distribution of income, fragile economic growth, a population in continuous growth, and increasing environmental deterioration. On the positive side, there has been a reduction of inflation, a reactivation of economic growth, elimination of military dictatorships, and growing regional economic cooperation. The changes taking place in Latin America today diverge from its recent historical trajectory, and lead to an uncertain future.

The current situation seems to represent an "explosion of novelty." Globalization, the Knowledge Revolution, and the social and economic impacts of the diffusion of the new technology will probably operate as triggers of global and regional changes, with significant social, economic, cultural, and environmental consequences. The resulting new directions in principle could lead to a worsening but also to an improvement of the situation in relation to past trends. In particular, the ecological future of Latin America, in terms of natural resources as well as of habitability, will depend on the social options adopted in the region in the turbulent context of the current world situation.

The ecological future of Latin America is directly linked to the large social options of the region more than it is to the search of new knowledge and new techniques for ecosystem management, although those are also necessary.

Several socioeconomic scenarios leading to alternative futures of the world and region are open. In each one of them it is possible to identify possibilities for the improvement of the management and conservation of the environmental resources. The opportunities and restrictions vary strongly between the different scenarios.

The countries of Latin America need to define and implement new development strategies allowing them to incorporate the opportunities implicit in the economic transformations and the new technologies without paying enormous social, economic, and ecological costs. In other words: Latin America must implement its own "knowledge revolution", shifting away from resource-intensive sectors and forms of production, into knowledge-intensive products and technologies.

However such strategies, in order to be viable, must be socially, economically, and ecologically sustainable in the long term. In economic terms, new forms of

[17] The effect of reduction in carbon emissions in the sustainable scenario described above is a good example. Reforestation and rehabilitation of degraded lands, besides contributing to mitigate global climatic change, have social and environmental value by themselves. Other things being equal, they are therefore preferable to actions that would be useful only if the phenomenon of global warming is really confirmed.

property rights for knowledge are needed if innovation and knowledge-intensive sectors and products are to flourish. The aim is to achieve a society having as basic attributes participation of the population in the creation and diffusion of knowledge and decisions, increased equity in the access to knowledge, and an intrinsic compatibility with its environment. This is a much bigger challenge, and of different nature, than developing economic competitiveness at the international level.

These new national and regional development strategies will have to be solidly based on science and technology, with priorities and goals established by the countries of the region. This is indispensable due to the fact that the knowledge-intensive sectors are those that contain most of the dynamics of the world economy today, and in many ways characterize the current globalization process.

The analysis presented here shows that within this context, trade liberalization may be desirable. In the context of resource-intensive industrialization, however, liberalization is not necessarily or automatically beneficial for all countries, and is certainly not a recipe for sustainable development[18]. This contrasts with the position of the international organizations that maintain that economic globalization will be beneficial for the South and that developing countries should therefore accelerate their total integration in the global economy through an extensive and fast liberalization based on resource extraction and exports, and the enhancement of the role of the market, reducing the function of the state to the maintenance of an environment appropriate for the flowering of business and the functioning of competitive markets.

From the environmental and social points of view, it appears that the countries of Latin America should carefully define policies for selective integration to the world market, instead of a fast unconditional opening of the economies.

Besides the inherent difficulties to the definition of the strategic integration required in different countries of the region, it is obvious that the current international tendencies represent a strong obstacle for any strategy endogenously defined (particularly for the poorest, but also for the strongest countries). However, even within the limited degrees of freedom allowed by the current constraints, it should be possible to adopt strategic directions favoring an orientation toward a more sustainable development in the region.

When the currents are strong and turbulent, there are at least two possible strategies: that of the transatlantic ship, whose power and robustness confront the frenzy of the elements, and that of the kayak, whose agility allows it to negotiate the fast waters on the basis of know-how and high capacity of reaction. The required skills and the actions appropriate to drive the two vessels are very different. It is possible that the countries of Latin America and the Caribbean should concentrate on the kayak metaphor at the national level, and in the transatlantic metaphor at the

[18] Some analysts even maintain that globalization and the liberalization policies, and the dismantling of the state, have failed in the industrial countries as well as in the emergent Asian economies, and that the fast growth of the latter was due to the application of policies precisely opposed to those recommended by the advocates of total liberalization (The South Centre, 1996). The IMF recognized, after the crash of the Asian markets in 1998, that the liberalization models have been pushed too far in Asia.

regional level (or even the extra-regional one, leaning on strategic alliances and real international cooperation).

In synthesis:

- Globalization is a multidimensional process that greatly exceeds the economic dimensions of free trade and the neoliberal position.
- While markets are an important economic institution and an engine of growth, an unrestricted opening of the economies to the international market is not by itself conducive to sustainable development. The free play of market forces can spontaneously generate asymmetries and inequalities, and under current conditions is unable to guarantee ecological sustainability.
- Some of the effects (both positive and negative) of globalization and of the opening of the economies are already detectable and are being documented.
- Globalization exhibits an essential ambivalence with potentially very positive and negative potentials. Therefore it is necessary to guide the globalization processes according to criteria of strategic importance.
- The globalization process should be associated with ethics, responsibility, and a governance system. The possibility that the global system falls into a scenario of generalized "barbarization" (Gallopín & Raskin, 1998) may otherwise be realized.
- The environmental sustainability of development poses specific questions for the strategies to cope with globalization (for example, the role of the different time horizons in economic decisions and in ecological processes).
- The current situation in Latin America and the Caribbean vis-à-vis the globalization process is weak. Under those conditions, the possible strategies are:
    - To simply drift with the currents (which seems to be the present attitude). This, for the reasons previously presented, could be suicidal. At best it can only benefit a minority, excluding most of the population of the region.
    - To navigate creatively: the strategy of the kayak—not synonymous with letting go, taking advantage of the force of the flows but at the same time seeding the germs of the desired future. This is possible only if a long-term vision is adopted, a systemic and multi-causal perspective, directed to create a society intrinsically compatible with its environment. It implies measures coordinated in space and staged in time directed to give fruit at different time horizons (institutional in the short term, educational in the long term, etc.)[19].
- And above all, more than to try to tighten all nuts and bolts, it is necessary to strengthen the societal and ecological sources of renewal, and the generalized societal capacity to respond to change, even to unexpected change.

[19] A third strategy would be to try to change, from Latin America and the Caribbean, the global system—the transatlantic boat. However, this is probably unfeasible. If possible at all, it would depend on the capacity of the region to establish alliances with other international actors, and in the future development of solid regional cooperation.

Economic globalization, in combination with the end of the Cold War and the potential of the Knowledge Revolution, makes possible a qualitative jump in the history of the human civilization. This promise, however, may not be fulfilled, and the growing marginalization of large population masses in the South (and also in the North) anticipates a conflictive future.

It seems clear that there are no separate solutions, one for the North and one for the South. A global solution must be obtained that integrates the needs of the North and of the South, and leads to rapid economic progress and to the creation of new technology while fostering the conservation of the world's precious natural resources. A solution must be found that integrates the needs of the present and the future.

Literature Cited

Berry, A. 1995. The Social Challenge of the New Economic Era in Latin America. FOCAL/CIS Discussion Paper. Centre for International Studies, Toronto.

CDMA-ALC (Comisión de Desarrollo y Medio Ambiente de América Latina y El Caribe). 1990. Nuestra Propia Agenda. IDB/UNDP, Washington, D.C.

Chayanov, A. V. 1925. Published by the Moscow Institute of Scientific Research of Agricultural Economy. Spanish translation 1974. La organización de la unidad económica campesina. Nueva Visión, Buenos Aires.

Chichilnisky, G. 1993. North South trade, property rights and the dynamics of renewable resources. Structural Change and Economic Dynamics [Oxford University Press] 4(2): 219–248.

Chichilnisky, G. 1994a. North South Trade and the Global Environment. American Economic Review 84(4): 851–874.

Chichilnisky, G. 1994b. Traditional comparative advantages vs. increasing returns to scale: NAFTA and the WTO. CHECK *In* M. di Matteo Baldassarri & R. Mundel (editors), International Problems of Economic Interdependence St. Martin Press.

Chichilnisky, G. 1995. Strategies for trade liberalization in the Americas. Pp. 165–188 *in* Trade Liberalization in the Western Hemisphere. InterAmerican Development Bank, United Nations, Economic Comission for Latin America and the Caribbean (ECLAC) Washington, D.C.

Chichilnisky, G. 1995–1996. The economic value of the Earth's resources. Invited Perspectives article in Trends in Ecology and Evolution (TREE) 135–140.

Chichilnisky, G. 1996. Trade regimes and GATT: Resource intensive vs. knowledge intensive growth. Economic Systems 20: 147–181.

Chichilnisky, G. 1997. Ecology and the Knowledge Revolution. Plenary Lecture at the National Academy of Sciences Second Forum on Biodiversity, "Nature and Human Society," 27–29 October 1997, Washington, D.C., published in the Proceedings (P. Raven, editor).

Chichilnisky, G. 1998. The Knowledge Revolution: Its impact on consumption and resource use. *In* S. Fukuda-Parr (editor), 1998 Human Development Report United Nations Development Program, New York.

Chichilnisky, G. & G. Heal. 1998. Economic returns from the biosphere. Nature 391: 629–631.

Daly, H. & R. Goodland. 1994. An ecological-economic assessment of deregulation of international commerce under GATT. Ecological Economics 9: 73–92.

De Bellevue, E. B., E. Hitzel, K. Cline, J. A. Benítez, J. Ramos-Miranda & O. Segura. 1994. The North American Free Trade Agreement: An ecological-economic synthesis for the United States and Mexico. Ecological Economics 9: 53–71.

Dourojeanni, M. J. 1982. Renewable Natural Resources of Latin America and the Caribbean: Situation and Trends. World Wildlife Fund, Washington, D.C.

ECLAC (Economic Commission for Latin America and the Caribbean). 1995. Panorama Social de América Latina. CEPAL, Santiago, Chile.

ECLAC. 2000. Equidad, desarrollo y ciudadanía. CEPAL, Santiago, Chile.

Ekins, P., C. Folke & R. Constanza. 1994. Trade, environment and development: The issues in perspective. Ecological Economics 9: 1–12.

FAO-FNUAP-IIASA. 1984. Capacidades potenciales de carga demográfica de las tierras del mundo en desarrollo. FAO, Rome.

Furtado, C. 1984. O futuro da América Latina. Pp. 67–72 *in* Seminario. A Crise Presente e o Futuro da América Latina. SP, 1984. Anais. SP, CESP, 1985.

Gallopín, G. C. 1980. Development and environment: An illustrative model. Journal of Policy Modelling 2: 239–254.

Gallopín, G. C. 1990a. Rapport Ecologique sur Notre Planète. Relations N1 564: 247–250.

Gallopín, G. C. 1990b. The environmental sustainability of development and technological change in Latin America and the Caribbean. LC/L. 577 (Sem.56/2); 8 August 1990; United Nations Economic Commission for Latin America and the Caribbean - ECLAC-; Santiago de Chile.

Gallopín, G. C. 1991. Human dimensions on global change: Linking the global and the local processes. International Social Science Journal 130: 707–718.

Gallopín, G. C. 1992. Science, technology and the ecological future of Latin America. World Development 20(10): 1391–1400.

Gallopín, G. C. 1994. Impoverishment and Sustainable Development. A Systems Approach. International Institute of Sustainable Development, Winnipeg, Canada.

Gallopín, G. C. (Editor). 1995. El Futuro Ecológico de un Continente. Una Visión Prospectiva de la América Latina. United Nations University Press and Fondo de Cultura Económica, Mexico, D.F. Two volumes.

Gallopín, G. C. 1995a. Medio ambiente, desarrollo y cambio tecnológico en la América Latina. Pp. 483–539 *in* G. C. Gallopín (editor), El Futuro Ecológico de un Continente. Una Visión Prospectiva de la América Latina. United Nations University Press and Fondo de Cultura Económica, Mexico, D.F. Volume II.

Gallopín, G. C. & C. A. Barrera. 1980. A Semi-quantitative mathematical model of the interactions between ecological and socio-economic factors in the Chaco Province (Argentina). Manuscript, Grupo de Análisis de Sistemas Ecológicos. S.C. Bariloche, Argentina.

Gallopín, G. C. & P. Raskin. 1998. Windows on the Future: Global Scenarios & Sustainability. Environment 40(3): 6–11 & 26–31.

Gallopín, G. C. & M. Winograd. 1992. Obstáculos y oportunidades para el desarrollo sustentable en América Latina. Pp. 1–30 *in* Fundación MAPFRE, Problemática futura del Medio Ambiente en América Latina. Editorial MAPFRE, Madrid.

Gallopín, G. C. & M. Winograd. 1995. Ecological prospective for tropical Latin America. Pp. 13–44 *in* T. Nishizawa & J. I. Uitto (editors), The Fragile Tropics of Latin America: Sustainable Management of Changing Environments. United Nations University Press, Tokyo.

Gallopín, G. C., M. Winograd. & I. Gómez. 1991. Ambiente y Desarrollo en América Latina y el Caribe: Problemas, Oportunidades, y Prioridades. Grupo de Análisis de Sistemas Ecológicos, S.C. Bariloche, Río Negro, Argentina.

Gómez, I. A. & G. C. Gallopín. 1995. Potencial agrícola de la América Latina. Pp. 497–547 *in* G. C. Gallopín (editor), El Futuro Ecológico de un Continente. Una Visión Prospectiva de la América Latina. United Nations University Press and Fondo de Cultura Económica, Mexico, D.F. Volume I.

Grossman, R. & A. Krueger. 1991. Environmental Impacts of a North American Free Trade Agreement. National Bureau of Economic Research WP 3914, Harvard University, Cambridge Massachusetts.

Herrera, A. O. 1986. The new technological wave and the developing countries: Problems and options. Pp. 140–153 *in* E. MacLeod (editor), Technology and the Human Prospects. Frances Pinter, London.

Kriström, B. 1996. Income Elasticity of the Demand for Environmental Goods and Services. Working Paper, Swedish Agricultural University, Umea.

Maletta, H. 1988. Macroeconomic Constraints on Soil Conservation in Andean Peasant Agriculture. Report to the Grupo de Análisis de Sistemas Ecológicos, S.C. Bariloche, Argentina.

Nochteff, H. 1987. Revolución Tecnológica, Autonomía Nacional, y Democracia. Monogr. e Inf. de Investigación N1 59. FLACSO, Buenos Aires.

Pérez, C. 1986. Las nuevas tecnologías: Una visión de conjunto. Pp. 44–89 *in* C. Ominami (editor), La Tercera Revolución Industrial. Impactos Internacionales del Actual Viraje Tecnológico. Grupo Editor Latinoamericano, Buenos Aires.

PNUD (Programa de las Naciones Unidas para el Desarrollo; United Nations Development Programme-UNDP-). 1989. Documento Técnico y Declaración Regional sobre la Pobreza. PNUD, Bogotá, Colombia.

PNUMA-AECI-MOPU (Programa de las Naciones Unidas para el Medio Ambiente–PNUMA-, Agencia Española de Cooperación Internacional–AECI- y Ministerio de Obras Públicas y Urbanismo–MOPU). 1990. Desarrollo y Medio Ambiente en América Latina y el Caribe: Una visión evolutiva. MOPU, Madrid, España.

Repetto, R. 1993. Trade and the Environment: Achieving Complementarities and Avoiding Conflicts. WRI. Washington, D.C.

Røpke, I. 1994. Trade, development and sustainability—A critical assessment of the "free trade dogma." Ecological Economics 9: 13–22.

Runnalls, D. & A. Cosbey. 1992. Trade and Sustainable Development: A Survey of the Issues and a New Research Agenda. International Institute for Sustainable Development, Winnipeg.

Schaper, M. 1999. Impactos ambientales de los cambios en la estructura exportadora en nueve países de América Latina y el Caribe: 1980–1995. Serie Medio ambiente y desarollo No. 9, Comisión Económica para América Latina y el Caribe (CEPAL), Santiago, Chile.

Schwartz, P. 1991. The Art of the Long View. Doubleday, New York.

Schneider, S. H. 1989. The Greenhouse Effect: Science and policy. Science 243: 771–781.

Sedjo, R.A. 1989. Forests: A tool to moderate global warming? Environment 31(1): 14–20.

Silva, J. E. da (Editor). 1994. Consideraçoes sobre o uso actual e potencial das terras do Cerrado sob o aspecto da sustentabilidade agricola. *In:* Strategies for Sustainable Agricultural Land Use in the Lowland Savannas of South America (SSALLSSA). Annex I. Country Documents. Project Proposal, CIAT, Cali.

South Centre, The. 1996. Liberalization and globalization. The issues at stake for the South and for UNCTAD. South Letter N1 25, Vols. 1 & 2: 2–7.

Sunkel, O. & N. Gligo (Editors). 1980. Estilos de desarrollo y medio ambiente en la América Latina. Fondo de Cultura Económica, México. Two volumes.

Tabatabai, H. 1995. Poverty and inequality in developing countries: A review of evidence. *In:* G. Rodgers & R. Van der Hoeven (editors), The Poverty Agenda: Trends and Policy Options. ILO, Geneva.

The World Bank. 2000. World Development Report 2000/2001. Attacking Poverty. Oxford University Press, Oxford.

Thurow, L. 1996. The Future of Capitalism: How Today's Economic Forces Shape Tomorrow's World. William Morrow, New York.

United Nations. 1994. World Population 1994. Population Division. Department for Economic and Social Information and Policy Analysis; ST/ESA/SER.A/142. New York.

UNDP (United Nations Development Programme). 1996. Human Development Report 1996. Oxford Univ. Press, New York.

UNEP (United Nations Environment Programme). 1999. Global Environment Outlook 2000. Earthscan, London.

Winograd, M. 1995. Simulación del uso de tierras: Escenarios predominante y sostenible. Pp. 371–482 *in* G. C. Gallopín (editor), El Futuro Ecológico de un Continente. Una Visión Prospectiva de la América Latina. United Nations University Press and Fondo de Cultura Económica, Mexico, D.F. Volume I.

Winograd, M. 1995a. Indicadores ambientales para Latinoamérica y el Caribe: Hacia la Sustenta-bilidad en el Uso de Tierras. IICA, San José, Costa Rica.

Winograd, M. 1995b. Anexo de los Capítulos 1 y 2. Clasificación de los Grandes Ecosistemas por Zonas de Vida. Pp. 164–180 and 371–482 *in* G. C. Gallopín (editor), El Futuro Ecológico de un Continente. Una Visión Prospectiva de la América Latina. United Nations University Press and Fondo de Cultura Económica, Mexico, D.F. Volume I.

WRI (World Resources Institute). 1996. World Resources 1996–97. WRI/UNEP/UNDP/WB. Oxford Univ. Press, New York.

Property Rights of Public Natural Resources: a New Legal Alternative for Developing Countries

Fernando Gabriel Irrera,
Mariano Javier Oneto,
and Jorge Rabinovich
Departamento de Investigacion,
Universidad de Belgrano,
Zabala y Villanueva,
1428 Buenos Aires, Argentina

Abstract

In many developing countries institutional and administrative constraints may hinder the sustainable use of natural resources. We analyze those constraints of legal nature. Although most of the experience is drawn from Argentina, and despite the differences in legislation and administration among countries, we believe that they can be generalized to a variety of situations in Latin America and the Caribbean. For simplicity, we consider a variety of situations as representative of the "South," a term we will use with that sense. We examine those goods that constitute the *public domain of the State*, and in particular natural resources, and we look into the judicial configuration that entitles exploitation and at the same time guarantees their care. This has two connotations: one is the care of resources for their preservation and maintenance of their natural renewal capacity; the other is that this care should be compatible with adequate economic profits as to be attractive as a business or, when used for subsistence—as in the case of rural people—to provide an adequate living standard for a household. These two conditions imply a sustainability approach, and a modification of the historical tradition of obtaining earnings without internalizing the costs of nature as provider and/or recipient of their use, that is to say, to avoid the transfer of potential negative externalities to the society. We propose the figure of the concession of use for public natural resources to individuals or firms or communal groups. This figure transfers to its holders a sense of "property" that satisfies a needed juridical and economic security; the right to use is endorsed by clauses

of rescission of the concession contract without fair cause, which would force the State to provide a compensation of the damages caused to the concessionaire. On the other hand, it leaves a safeguard on the property right of the State and of the Provinces over their resources, since the right of the concessionaire represents a "property" for the use and not of the resources itself.

Introduction

In many developing countries institutional and administrative constraints may hinder the sustainable use of natural resources (Rabinovich, 2001 this book). In what follows we will analyze those constraints of a legal nature. Although most of the experience is drawn from Argentina, it may be applicable to some other Latin American countries as well. For simplicity, we will consider the variety of situations as representative of the "South," a term we will use in this sense.

Legal and administrative constraints are usually complicated because of the critical economic and social conditions in many states of the South. Thus it is difficult for a broken state to fulfill its duties of caring for the environment. The analysis and proposal here elaborated are based on the idea of the conversion of natural resources into a business for both the State and the private sector, in order to foster the rehabilitation and maintenance of the natural resource base.

We analyze here the goods that constitute the *public property of the State*, and in particular natural resources, and we look into the judicial configuration that entitles their exploitation and at the same time guarantees their care. This has two connotations: one is the care of the resources for their preservation and maintenance of their natural renewal capacity; the other is that this care should be compatible with adequate economic profits so as to be attractive to a business or, when used for subsistence—as in the case of rural people—to provide an adequate living standard for the household. These two conditions imply a sustainability approach, and a modification of the historical tradition of obtaining earnings without internalizing the costs of nature as provider and/or recipient of their use; that is to say, to avoid the transfer of potential negative externalities to the society.

To be able to outline and justify an alternative to the sustainable use of the natural resources in the South, we should review different legal definitions for a potentially applicable configuration.

Goods of the Public Property of the State

Consistent with the federal spirit prevailing in Argentina, article 2339 of the Civil Code provides the definition "...things are *public* goods of the general State that forms the Nation, or of provincial States that she is composed of, according to

the distribution of the powers made by the National Constitution; or they are *private* goods of the general State or of the provincial States" [italics added for emphasis].

The National Constitution of Argentina, as reformed in 1994, prescribes, in its article 124, that "the original domain of the existing natural resources in its territory" corresponds to the Provinces, thereby respecting the Provinces' original ownership upon its resources (Dallavía, 1994). Article 2340 of Argentina's Civil Code establishes that the public goods of the State are: the territorial seas, the interior seas, bays, ports, rivers, river beds and their underground waters, marine beaches, the navigable lakes and their channels, the islands when they do not belong to individuals, the streets, plazas, roads, channels, bridges and any public work, the ruins and archaeological and paleontological sites of scientific interest (Pigretti, 1987).

With regard to the use of the public goods of the State, the Argentinean Civil Code (article 2341) establishes that individuals can use them, but are subjected to the dispositions of this Code and to the general or local ordinances; thus the State can modify ways in which inhabitants can use public goods. For example, if a lake is accessible for the use and enjoyment of the general population, but later it becomes a national park, the use and enjoyment will, from that moment on, be governed by the rules of the national park.

The concept of private domain of the State is equivalent to the domain of civil rights except that the proprietor is the State; this implies that, in spite of the norms of public right that govern it, its assimilation to the private domain of the individuals is obvious and therefore, the things that comprise the private domain of the State can be sold, levied, etc.

On the other hand, the category of public domain of the State is evident: it involves the judicial authority of the State on everything enunciated in article 2340 of the Civil Code of Argentina, for use and benefit of the people. The public domain of the State has nothing to do with the codes of private rights, for its authority is a materialization of the public power of the State, that is, of its sovereignty.

A precise identification of the public goods of the State is essential because Argentina's National Constitution omitted listing which of the public goods belong to the provincial States (despite the current constitutional amendment of article 124; Bustamante Alsina, 1995). This omission is covered by the Civil Code that enunciates which are the public goods of the State (article 2340) and makes it clear that all those public goods correspond to the Provinces if located within their territories (article 2339).

For example, it corresponds to the Provinces the public domain of the rivers and, on the basis of this right, the Provinces have established regulatory laws on their water patrimony, although there are cases where the Nation has priority because "the sailing of the interior rivers of the Nation is free for all flags, subjected only to the regulations dictated by the national authority" (article 26 of the Argentina's National Constitution).

Domain and Property

The Civil Code of Argentina defines domain as "...the real state by virtue of which a thing is subjected to the will and the action of a person" (article 2506). To this definition we can add that the domain should be exclusive because, otherwise, we would be in the presence of another real state: co-ownership.

On the other hand, for the Civil Code the concept of property is wider than that of domain since "... the property of a credit goes to the transferee by effect of the transference..." (article 1457) and "... the property of a credit is not transferable to the assignee..." (article 1459). We conclude that property is the "gender" and that, when the property is an object, a "species" is established, which is the domain.

If property rights over certain goods of the public property of the State have been granted to a person, firm, or cooperative, it does not mean that domain has been granted. This distinction is an essential ingredient to our proposal, since it allows the consideration of granting a property right (and becoming temporarily part of an individual's patrimony) despite the State conserving its domain.

Possible Rights Over Public Goods

Given the importance of property rights, we will first analyze which are the possible rights that people can be entitled to in relation to the public goods of the State or of the rovinces; we will examine the different legal configurations that *cannot* be applied according to Argentina's legislation, and in doing so we will try to arrive at one that would allow it.

Right law

The right law represents a group of faculties that corresponds to individuals, and that can be exercised by them to make effective their juridical power. It consists of four elements: (a) the person or individual, entitled to a right and whose interest is protected; (b) the object or person on which the individual exercises its power; (c) the cause, or act or juridical fact that gives birth to the subjective right; and (d) the passive subject, that is, the person that is subjected to the inherent obligation of the property's right. The latter implies that all right law means is a relationship among individuals that results in that when a person is entitled to a right another person is subjected to an obligation.

The conformation of a figure that grants a property right on goods (natural resources in this case) that belong to the State or to the Provinces can be considered as a right law if it fulfills the above-described characteristics. We have only to systematize appropriately the cause that would give birth to this right law because real state (property, perfect usufruct) can only be created by law and, being goods of the public domain of the State, a norm should also exist to allow granting such a right. These real state (property, perfect usufruct) are subjective rights *erga omnes*, that is to say, it corresponds them the obligation of abstention of all the other fellows.

The domain

This figure is not applicable for a property right over natural resources because the domain over public goods belongs to the State or the Provinces.

The perfect usufruct

This is the real state of use and enjoyment of other people's things. It follows in hierarchy to the domain, for the usufructuary (*holder of the right of perfect usufruct*) can only use and enjoy the thing object of perfect usufruct but does not have the faculty to sell or dispose of it. Furthermore, the use and enjoyment of the thing is not as wide as that of the owner since it cannot physically alter it nor change its destination (for example, it cannot transform a shepherding field in a crop field). Respecting these limitations, the entirety of the use and of the fruits coming from the thing, belong to the usufructuary (Arean, 1986; Gatti, 1984).

An important attribute of the perfect usufruct figure is that it is temporary, its maximum term being that of the life of the usufructuary, and it is not transmitted to its heirs. If natural resources are given in perfect usufruct, with this figure the State never loses the domain over them.

Article 2839 of Argentina's Civil Code establishes that perfect usufruct cannot be constituted over goods of the public domain of the State or of the Provinces, but it permits that perfect usufruct can be constituted over goods that belong to the private domain of the State or of the Provinces; however, this can only be done by means of a special law that authorizes such an affectation. Consequently, to be able to carry out a usufruct of public goods of the State or of the Provinces, a law that modifies Argentina's Civil Code should be passed (and this can only be done by the National Congress) eliminating this prohibition. This law should also contain, as a warrant of the adequate care of natural resources, the rules that regulate this exercise, that is to say, the limitations to the use and enjoyment of these goods in benefit of the people at large.

In summary, the figure of real state is not well adapted for the economic use of the public goods of the State or of the Provinces, unless a modification of the usufruct is carried out, a figure that looks to be the most promising for clearly defined property rights.

Concession (Grant) of Use of the Natural Resources

Within the administrative right we have the figure the "concession of use" that is different from the "permission of use." While the latter is a mere tolerance on the part of the Public Administration, the "concession of use" implies for its holder a complete right law that cannot be revoked at any moment without compensation, as it happens with the permission of use (Marienhoff, 1996).

In the Argentinean legislation the concept of concession of natural resources does not exist, since this modality belongs to the European continental juridical

system. Only recently, in the 1990s, the idea of the privatization of a great part of the private goods of the State was revitalized in Argentina, leading to a privatization wave under the figure of public concessions, involving most public services (light, gas, mail, telephones, drinking water) that up to then was covered by a "paternalistic" state. However, this figure did not become extensive to the use of the natural resources, as we here propose, and that would be coherent with most current political, social, and economic ideas.

Dromi (1995) claimed that "...the public domain is a group of goods belonging to a public figure that, as they benefit the people at large, is subjected to a special juridical régime of public right" and that "... the goods under this domain are not *res nullius*, goods without owners, things of nobody, but rather the subject holder of the goods under public domain, is the state, or the public figure." According to this interpretation, as the natural resources belong to the original domain of the Provinces they are not *res nullius*, but have the national or provincial states as proprietor.

Although the goods of the public domain of the State do not prescribe and cannot be sold, Dromi (1995) claimed "...that the goods of public domain are inalienable doesn't mean that they are totally outside of the juridical trade, they can be object of special rights of use, granted or acquired by means of the forms recognized by administrative laws." Consequently, the natural resources (not being *res nullius* and having the Provinces as proprietors), in spite of being inalienable they can be subjected to be given in concession, a figure that we propose should be enacted by law so that the use of the natural resources can be given to individuals or firms, for a fixed time, and with specific sustainable rules.

According to Dromi (1995), the nature of the concession right constitutes a right law, that is, a property of the licensee (concessionaire) in the constitutional sense of the term, which grants the licensee (concessionaire) a juridical security not available at present times in connection with natural resources. The juridical security is achieved through a concession (grant) contract that is not a capricious tool in the hands of the State; possible modifications to the concession (grant) contract have to be reasonable and justified, and in the case they would take place (since it is an obligation of the State to safeguard the public interest), the concessionaire (licensee) will be entitled to a fair compensation. This figure of a concession of natural resources implies a warranty to the concessionaire that it will not be subjected to the arbitrary changes of policies with changes in the administration. The juridical security granted by this figure to individuals and firms with regard to the natural resources is an assurance for the development of the business.

This proposal of the concession of use should not be defined from the vision made from the private right, but from a wider perspective, since we are in the field of the public right where the interest of the people at large is above all other things. We have found in the Argentinean legislation related to the regulation of the use of various natural resources, indications that open the way for the juridical figure here proposed. The following are some examples:

Water resources

Article 2350 of Argentina's Civil Code prescribes that "...the springs that are born and die inside a given estate, belong, in property, use and enjoyment, to the owner of the estate"; it also sustains that non-navigable lakes do not belong to the public domain.

Wildlife resources

Law 22421 enacts, in its article 8, that "...conforming national and provincial legal dispositions and norms, the owner of a state will be able to use the wildlife that inhabits it transitorily or permanently, and shall protect it and rationally limit its use to assure its conservation." Here the State, in spite of not granting property to an individual or firm, demands that its behavior should be as if the wildlife would really be of the individual's or firm's domain.

Forest resources

Law 13273, related to the common forest régime, grants to the occupant or owner of the forests their use subjected to the elaboration of a rational management plan that has to be presented to the pertinent authority (Pigretti, 1987).

Mineral resource

The amendment to Argentina's Code of Mining in November 1995 stimulated in our country an increase of mining exploitations because it allowed mining companies to exploit the mines freely, without being subjected but to three rules: security, police, and environmental conservation.

Usefulness of the figure of "concession—grant of use"

From the above arguments we conclude that in the Argentinean legislation we can find timid "privatization intents," although with some differences with the juridical concept here proposed of the concession of the natural resources to individuals, firms, or communal groups. This step should be given within an appropriate social, political, economic, and juridical framework, and this figure should have as its supreme purpose that of the benefit of the people at large, and that of a "healthy" environment.

Conclusions

The proposed figure of the concession of use for public natural resources to individuals or firms or communal groups transfers to its holders a sense of "property" (*sensu* Clerc, 1982) that satisfies the juridical and economic security that was needed; the right to use is endorsed by clauses of rescission of the concession

contract without fair cause that would force the State to compensate for damages caused to the concessionaire-licensee.

On the other hand, it leaves a safeguard on the property right of the State and of the Provinces over their resources, since the right of the concessionaire represents a "property" for the use and not of the resource in itself.

The figure of the proposed concession would require of a national "frame" law or, alternatively, of direct provincial laws that regulate the concession of the natural resources to put special emphasis, not in the business aspects (that is already implicit in its nature) but in an explicit protection of the natural resources assigning an administrative organism the control of the sustainable exploitation of the different types of resources. The greater the concession (in number of resources or in extension) the bigger should be the control put by the State in the safeguard of the resources object of the concession of use.

The figure here proposed could be criticized from the point of view that the enactment of laws that regulate the concession of the use of natural resources would arise from the area of the administrative rights, which is usually restricted to the specification of the conditions and rules applicable to the concessions. We agree that to give in concession a lake, a river, or a forest with complex relationships between plant and animals species deserves special care; thus we do not believe that their regulation should remain solely in the administrative responsibility, but rather it should be the fruit of a detailed interdisciplinary analysis translated in a regulation of higher and obligatory nature. "To make business taking care of the atmosphere" is from now on, perhaps, the slogan of the new knowledge civilization.

Acknowledgments

We thank Dr. Roberto J. Vernengo, Universidad de Buenos Aires.

Literature Cited

Arean, B. 1986. Curso de los Derechos Reales. Editorial Abeledo Perrot.

Bustamante Alsina, J. 1995. Derecho Ambiental. Editorial Abeledo Perrot.

Clerc, C. M. 1982. El Derecho de Dominio y sus Modos de Adquisición. Editorial Abaco.

Dallavía, A. 1994. Constitución de la Nación Argentina Comentada. Editora Platense.

Dromi, R. 1995. Derecho Administrativo. Ediciones Ciudad Argentina, Buenos Aires.

Gatti, E. 1984. Teoría General de los Derechos Reales. Editorial Abeledo Perrot.

Marienhoff, M. 1996. Permiso Especial de Uso de Bienes del Dominio Público. Editorial Abeledo Perrot.

Pigretti, E. A. 1987. Derecho de los Recursos Naturales. Editorial La Ley.

Rabinovich, J. 2001. Environment and economics from the perspective of the South. *In* Managing Human-Dominated Ecosystems. Monogr. Syst. Bot. Missouri Bot. Gard.

Economics and Environment in Developing Countries: Non-Traditional Environmental Policy Instruments

Carlos Galperín and Jorge Rabinovich
Departamento de Investigación
Universidad de Belgrano
Zabala 1837
(1426) Buenos Aires, Argentina
cgalpe@yahoo.com and rabinovi@netverk.com.ar

Abstract

Different alternative instruments applicable to developing countries are analyzed. The main regulatory and economic instruments are summarized, and their weaknesses and strengths are discussed, concluding that economic instruments complemented with new enforcement and dissuasion instruments improve effectiveness and efficiency of environmental policies, even in developing countries. Also analyzed are policy implementation conflicts in these countries, mainly between the private sector and the government, as well as among different governmental organizations and levels. A discussion trying to answer the following question is presented: Do changes in contemporary society tend toward more efficient policies to solve environmental problems?

1. Introduction

Reaching a sustainable development path requires appropriate management of the natural environment, both as a source of goods and as a recipient of wastes. In the latter case, pollution is defined as "alteration of the characteristics of the environment in such a way as to make it unfit for specific purposes" (Palange & Zavala, 1987).

Economic analysis asserts that the solution to this problem requires that economic agents take environmental externalities into account in the decision-mak-

ing process in order to modify their behavior. Natural sciences information is essential to identify the extension and magnitude of the problem, but policy aspects concern mainly social sciences that "...can only be addressed through changing human behavior—altering the demand for environmental services and changing and controlling their supply" (Helm & Pearce, 1990). The debate on environmental policy alternatives lies in this framework.

Examples of alternative instruments are: (i) regulations—also called command and control instruments (CAC); (ii) economic instruments, (iii) dissuasive instruments, and (iv) different enforcement mechanisms. These instruments were originally defined for developed countries. Are they applicable to developing countries?

Environmental policies in developing countries face particular problems, due to certain social, economic, and political conditions currently prevailing (Rabinovich, 2001 this volume): institutional weaknesses of regulatory agencies; limited budget for environmental policy; lack of good information about the environmental situation; intra- and intergovernment-agency jurisdictional overlap; standards that technological changes, trade liberalization, and deregulation made obsolete; limited public participation and weak enforcement capacity (Eskeland & Jimenez, 1992; Margulis, 1993; Bruce & Ellis, 1993; Banco Mundial, 1992). A good description for the Argentinean situation is given in the World Bank's 1995 report on Argentina.

In this paper we present different alternative instruments applicable to developing countries. We start with a summary of the main regulatory and economic instruments, complemented with new enforcement mechanisms and dissuasion instruments, and discuss their weaknesses and strengths. We then analyze policy implementation conflicts of developing countries, and we end with a discussion about the question: Do changes in contemporary society tend toward more efficient policies to solve environmental problems?

2. Regulatory Instruments

Regulatory instruments are "institutional measures aimed at directly influencing the environmental performance of polluters by the establishment and enforcement of laws and regulations prescribing objectives, standards and technologies polluters must comply with" (OECD, 1993). That is, a polluter can choose between complying or facing penalties (fines, temporary firm closure, prison) for not doing so. Effectiveness depends on the agencies' capacity to enforce and control the regulation. Although this definition is focused on pollution, the same instruments are applicable to many natural resource management activities that affect the environment.

The main advantage of the regulatory instruments lies in that, being the most frequently used, there is long experience and tradition behind them. Additionally, when there are potentially dangerous, irreversible consequences, e.g., in levels of toxic emissions, safe levels of control can be attained more directly and with more

accuracy through quantitative regulations than through a price adjustment mechanism, provided that adequate enforcement exists.

The weaknesses of the regulatory instruments are: (i) moral hazard and adverse selection problems generated by information asymmetries; (ii) the process of setting up standards and the performance of the controlling agencies are subjected to political pressures; (iii) from a dynamic point of view, regulations provide fewer incentives to use technological changes because they usually only prescribe a limit to emissions; and (iv) they are expensive to society, because they demand more resources for enforcement and because they do not consider firms' possibilities for a more efficient solution.

In relation to point (i), moral hazard refers to a situation where some endogenous variables are not easily observed by the regulator (e.g., hidden actions like a firm's effort to diminish emissions), and adverse selection arises if the regulatory agency has less information than the firm about some exogenous variables (e.g., hidden characteristics like feasibility of different technological alternatives); however, the latter is more relevant than the former.

Point (ii) is particularly important, and the problem of a regulatory agency's decisions (and even the rules that guide its fulfillment) as being influenced by the regulated firms is well known—particularly in CAC instruments, where regulations are set on a sector-by-sector basis. However, changes in variables exogenous to the agencies may modify the relative influence of vested interests: a greater environmental awareness level in society would increase the weight of public opinion and of environmental non-governmental organizations (NGOs) on standards and agency performance (Laffont & Tirole, 1993).

Even in developing countries, where these instruments are predominantly used, alternatives are being sought. Not only are their effectiveness and efficiency being questioned, but also tighter fiscal budget constraints encourage governments to look into less expensive methods of control, and even as a source of money to improve their fiscal condition.

3. Economic Instruments

One alternative is given by economic instruments, which provide market signals through relative price modification and through financial transfers, with the aim of influencing economic agents' behavior by impinging their cost and benefit estimations (Barde, 1994). Their main feature is that they allow each agent to choose how to answer to the economic incentive. Examples of economic instruments are taxes, subsidies, deposit-refund systems, and emission tradable permits.

These instruments have several advantages: (i) a better static efficiency, whereby each agent can design the least expensive solution according to his productive structure when faced with taxes or the tradable licenses, specially when the CAC approach is designed without any cost-effective consideration, as usually happens (Oates et al., 1989); (ii) a better dynamic efficiency, for they provide an incentive

to develop and use technological changes both in products and processes, since increasing efficiency reduces the cost of avoiding pollution and diminishes tax payments; (iii) the environmental agencies face smaller information costs than in direct regulation in relation to polluters' technical characteristics; (iv) a policy based on these instruments is less vulnerable to regulatory capture, because they are more transparent (OECD, 1996); (v) they are more flexible than standards, since it is easier to adjust a tax rate than the standard of a direct regulation; and (vi) except for subsidies, they are a source of financial resources that could be applied to environmental policy and research and development on technological alternatives.

In relation to point (iii), particularly in pollution cases, it is important to underline that in the design, implementation, and control of economic instruments, it is pollution reduction that matters and not how firms do it. This aspect, linked to the advantage of static efficiency (see point i), reflects the larger market capability to process disperse information and to allocate economic resources.

Point (vi) leads to the issue of the so-called double dividend. Environmental tax revenues (or revenues from auctioning tradable permits) can also be used to cut rates of pre-existing distortional taxes, such as on labor or capital, and then reduce the economic costs of environmental taxes or non-free emission permits (Repetto et al., 1992; OECD, 1996). But this revenue-recycling effect could be more than compensated for by a tax-interaction effect that points out that environmental taxes tend to magnify previous market factor distortions (Bovenberg & Goulder, 1996; Goulder, 1998).

On the other hand, these instruments present some weaknesses: (i) determination of tax rates is a complex matter; (ii) subsidies depend on available fiscal resources; (iii) in the case of permits, it is not easy to determine the number of allowable permits and to define the proper area of application; (iv) they do not work well if economic agents do not respond to incentives, as is usual in the case of firms belonging to the state; (v) it is more difficult to predict the amount of wastes that a firm might generate; (vi) in the case of very dangerous wastes, it seems safer to control directly the level of emissions (if adequate enforcement exists); and (vii) they are somehow resisted and in a certain way they leave firms with a right to decide how to pollute.

In general, governments resist implementing market-based incentives (Pearce & Warford, 1993). However, this is changing in developed and developing countries, because previous successful implementations in some countries facilitate imitation, and because conceptual foundations of public policies have been changing toward deregulation, integration of public policies that have the same objectives, and prevalence of preventive policies (OECD, 1989).

Economic instruments can be transformed into useful tools to solve many of the developing countries' environmental policy problems, e.g., (i) given governmental budget constraints, taxes may be earmarked to cover environmental policy costs; (ii) given the extremely scarce information available to regulatory agencies, economic instruments are less information-demanding; (iii) being more flexible, they

favor the adoption of new technologies that become available due to trade liberalization; and (iv) the classical institutional weaknesses of environmental agencies lose importance as tax implementation and enforcement fall on the shoulders of tax agencies, with more experienced and better administrative aptitudes.

However, there are still two remaining problems also shared by regulatory instruments: enforcement and implementation conflicts.

4. Dissuasive Instruments and Different Enforcement Mechanisms

These are an important aspect of all environmental policies, because independent of the instrument used, policies are faced with firms that do not always play by the rules or do not have total control over their levels of emissions. Thus, governments must not only set parameters for the chosen instrument, but must also design monitoring procedures and fines and penalties for violators (Cropper & Oates, 1992).

The economic point of view of enforcement considers that a person commits a transgression "if the expected utility to him exceeds the utility he could get by using his time and other resources at other activities" (Becker, 1968). This implies that a person will comply with the law as a function of the probability of being punished and the type of penalty, as well as of other variables such as his/her willingness to commit illegal acts, and the benefit from complying with the law versus the benefit from not doing so. If punishment levels and/or probability of being caught in a violation increase, his/her expected utility for committing the transgression will decrease and, in consequence, the number of illegal acts will decrease. This will also occur if, as Becker (1968) argued, education increases law-abidingness.

Environmental laws and regulations will be respected if the cost for doing so is less than the expected value of not abiding by them. Some of the factors that drive conforming to regulations are: (i) penalty level (fine, plant closure, prison); (ii) level of penalty enforcement; (iii) vigor and effectiveness of audit regimes; (iv) the risk of corporate image loss if discovered violating regulations; (v) reduction of sales for not respecting the law; (vi) attitudes toward the risk of being caught; and (vii) the "environmental conscience" level of potential polluters.

Governments, particularly in developing countries, usually act on the first three factors, although their results are not always successful. Thus this opens space for non-traditional enforcement alternatives based on the other factors. For instance, the probability of being caught increases when private actions, judicial or extrajudicial, are allowed, what is known as private enforcement mechanisms (Tietenberg, 1996). This implies the establishment of a legal framework where individuals or civil associations can initiate complaint actions (i.e., to lodge a complaint against polluters with the regulatory agency), oversight actions (i.e., to bring a legal action against the regulatory agency) or direct citizen suits (i.e., to bring actions against polluters based on diffuse rights), besides ordinary civil liability actions if they have been directly damaged by the pollution.

Private actions can complement public monitoring and auditing, thus increasing the probability of discovering violators, and as a consequence, increasing non-compliant costs to firms. These mechanisms are also attractive because the kind of motivation that makes individuals or civil associations undertake these actions, make them more resistant to lack of financial resources, bribery, or to become captured by the violator, typical problems in developing countries.

The negative part of this mechanism is its high information cost: fulfilling a legal action requires some kind of proof, which in turn implies that the private monitoring and auditing sector must have adequate technical resources and legal authorization. However, if regulations force waste reports to be made public, this reduces substantially the information cost (of course, regulations should also exist that set up penalties for hiding or falsifying information). Another important limitation is the slowness and inefficiency of the courts; however, this problem could be bypassed through mediation and arbitration, mechanisms now used in several Latin American countries.

Factors (iv) and (v) depend mainly on moral and social values that prevail in markets where potential polluters produce and sell their products. If consumers disapprove of pollution by buying less from polluter firms, those firms will try to improve their environmental behavior. Social values (those that society imposes on its members, and are only subjected to social sanctions) and moral values (those that require personal adherence) can be modified through environmental education, usually carried out by public and private schools, the media, and NGOs. North (1984) has analyzed the relation between moral and social values and the reduction in enforcement costs.

Dissuasion instruments maintain a close relationship with the improvement of the compliance level. These instruments aim at "internalizing environmental awareness and responsibility into individual decision-making by applying pressure and/or persuasion either directly or indirectly" (OECD, 1993), seeking that economic actors decide willingly not to behave as free-riders (Galperín, 1995).

Among the dissuasion instruments we find: (a) environmental education programs; (b) the proclamation of the benefits of preserving environmental quality as advertised by governments, NGOs, and chambers of commerce and industry; and (c) the publication in wide-circulation newspapers of the violators' names and public access to pollutant emission databases (e.g., the Toxic Release Inventory (TRI) in the U.S.; for an analysis of the TRI effects on firm and investor behavior, see Hamilton, 1995, and Konar & Cohen, 1997). Through these instruments social and moral values are influenced, impelling economic actors to seek more environmentally friendly alternatives, independently of the adequacy of laws and mechanisms of public enforcement. These dissuasion instruments reduce legal enforcement costs, of growing importance in developing countries with agencies lacking good monitoring and control powers (Galperín, 1995).

Green labels could be included in this category. These are marketing tools that require the existence of "green" consumers, with behavior influenced by prevail-

ing ruling social and moral values. This instrument will grow in importance as children who are now receiving environmental education begin influencing consumption patterns. In developing countries, green labels are important not only because of domestic-demand patterns, but also because of the foreign-demand patterns of importing countries.

Non-traditional instruments related to environmental education can also reduce enforcement costs by way of factors (vi) and (vii). They may produce that effect through: (a) affecting the potential polluter's attitude toward risk of being discovered; (b) inducing potential polluters to decide autonomously not to pollute; and (c) benefiting developing countries because the behavior of the local branches of transnational firms (with head offices located in developed countries) will be expected to follow strict environmental management all over the world (the existence of this mechanism in Argentina was studied by Chudnovsky et al., 1997).

With regard to (a), the reason is that the attitude toward the risk of being discovered and punished influences the extent of the auditing and control system and, consequently, on its cost: a more risk-adverse attitude would lead to a less expensive system (Becker, 1968; Polinsky & Shavell, 1979). With regard to (b), it is interesting to note that the United States Environmental Protection Agency (USEPA) argues that social and moral values are natural factors that favor compliance with the law, and that factors "such as the fundamental sense of social responsibility to obey the laws, and the belief that protection of the environment is a civic and personal duty...reflect widely held values in the United States today" (USEPA, 1990). In a similar sense, Cropper and Oates (1992) stated that "perhaps public opprobrium is a stronger disciplinary force than economists are typically inclined to believe."

An advantage of these non-traditional mechanisms is that a more effective enforcement reduces the importance of control feasibility evaluation at the time of choosing a set of environmental policy instruments. However, experience shows that they are not sufficient by themselves, although they can serve as a good complement to regulatory and economic instruments.

5. Implementation Conflicts

Once the most appropriate set of environmental policy instruments has been selected for a given situation, the implementation aspect becomes essential, for no matter how good the selected instrument is, a deficient implementation process can defeat its purposes.

We can distinguish different kinds of government conflicts that lead to implementation problems: between the government and the private sector target of the new instruments; among private sectors (some will lose and some will win); and within the government itself. The probability of a successful implementation depends crucially on how these conflicts are tackled.

5.1. Implementation conflicts between private sector and government

Any implementation strategy must be thought of not only as an approach to improve an environmental policy's effectiveness, but also to guarantee that a given instrument has the least negative impact on the economic activity, both for economic efficiency as well as for acceptability reasons. For instance, setting a predictable and stable instrument and announcing it in advance implies less disturbance on firms' investment plans, minimizing resistance by business and trade unions.

With this purpose in mind, we present below a model to show some of the problems that could result from the introduction of an emissions tax; we will then modify this model in order to present alternative solutions. It is developed for environmental taxes and for a water pollution problem, because it is a frequently used instrument and a fairly usual problem, although it can easily be adapted for a tradable permit system and/or a different type of environmental problem (air pollution, mismanagement of forest resources, etc.).

This tax is designed as a strategy to achieve a maximum permissible level of pollution and not to reach a Paretian optimum; that is, this tax is closer to Baumol's proposal than to a Pigouvian approach (for the shortcomings of the Pigouvian approach as an operational criterion, see Baumol, 1972).

Model I

Imagine the situation of a watershed with a river polluted by industrial effluents, and the problem cannot be solved with present policies. For simplicity, let us consider first only two actors: government (G), seeking a solution through the implementation of an emissions tax, and the polluters' sector (P). The only way for (P) to pay less tax is to decrease production in order to reduce pollution outflows. The traditional view asserts: a green tax will increase production costs, will decrease industry competitiveness, thus reduce industry profits and, consequently, a decline in production (and employment) in the polluting sectors. However, additional effects are: (a) firms' hindrance; (b) trade unions' opposition based on the risk of unemployment (with an eventual alliance with business sectors); and (c) opposition by the citizenry. When employment is a priority issue in the political agenda, the combination of these three effects usually makes tax implementation politically unviable.

Model II

To the above model we could incorporate other actors: economic sectors that supply environmental equipment and services, and other goods and services for environmental protection and management (OECD, 1992: the environment industry, EI), and those whose activity is negatively affected by the polluters (e.g., recreation activities, real estate, construction, and domestic water supply firms (A). In this new situation, consequences on (P) remain the same: reduced profits, and de-

creased production and labor demand. But if in order to pay less tax a firm decides to reduce emissions of a given pollutant, now it can demand goods and services of (EI), so that costs of (P) would increase not because of taxes but from buying equipment and services. Of course, pollution reduction diminishes negative externalities suffered by (A).

There is no consensus, from the theoretical point of view, on the question whether paying attention to environmental problems in a firms' management leads to an increase in production costs and less competitiveness (Palmer et al., 1995) or, to the contrary, whether it improves efficiency (Porter & van der Linde, 1995). In an empirical paper, Jaffe et al. (1995) concluded that real-world evidence is not enough to support either of the two hypotheses.

In this model government (G) faces a private sector divided between those who oppose the tax (P) and those who back it because they win with it (A and EI). It is a case of conflict between two groups, and its solution will depend on the relative strength of each. But, should the government intervene to balance a situation where there are winners and losers? There are reasons in favor and against intervention (OECD, 1996). Arguments in favor of intervention are: (i) losers may have such a lobby capability as to block tax implementation if it is not changed; and (ii) among the losers there are usually a substantial number of poor people (and public policy social equity principles would favor intervention). Arguments against intervention are: (i) it bears the image of a government sensitive to pressures by vested interests; (ii) the presence of winners and losers is common in any short-term policy: Paretian solutions are the exception and not the rule; and (iii) as pollution is socially disapproved, government would have to explain why to help polluter firms.

However, as political pressures are a reality in all countries, it is convenient to include a balance between different measures into the tax implementation strategy. These measures are of two types: (a) mitigation, if it involves tax structure adjustments in order to reduce or avoid negative consequences; or (b) compensation, if it refers to aid the losers. Compensation is considered advantageous over mitigation because it does not decrease tax effectiveness. Obviously, the former is *ex ante* and the latter is *ex post*. If we consider these measures, the model is modified once more and we are left with Model III.

Model III

Environmental taxes do not affect relative competitiveness of (P) if other taxes are reduced as a compensation and if technical and financial assistance is provided to decrease emissions. This assistance may consist of free technical advising but tied to the obligation of implementing some of the suggested solutions, and in government subsidies through a redistribution of the environmental taxes collected and/or preferential credits. As a complement, an anticipated announcement and a gradual implementation of the environmental tax will allow firms to plan ahead for their investments and modifications of their production structure, reducing the cost of complying. A similar result is obtained from the combination of taxes and volun-

tary agreements in pollution reduction: firms win time but they agree to change their environmental behavior, and the government increases the probability of a successful environmental policy.

When reducing taxes to help maintain the relative competitiveness of (P), this tax rebalancing diminishes the impact on firm costs and keeps Treasury fiscal revenues constant, avoiding criticism of a tax pressure increase. But it also faces some problems: (i) if the environmental tax is effective (pollution decreases and fiscal revenue falls), should tax compensation end?; (ii) as not all the sectors are affected equally, nor do all firms produce the same pollution damage, the discrimination by sector and by firm excessively increases administrative and tax enforcement costs. Thus, this kind of compensation is difficult and expensive to implement.

On the other hand, pollution control costs may be shared among those who pollute (P), those who benefit (A and EI), and the government (G). Four alternatives can be considered: (i) the burden is carried only by (P); (ii) the burden is carried only by (A) and (EI), and taxes collected are redistributed by the government among those firms that reduce pollution (those who gain with the reduction pay for the positive externality they receive); (iii) a combination of (i) and (ii): taxes to (P) and (A) and (EI); (iv) any of the first three alternatives, plus government subsidies financed from general taxes.

Model comparison and final considerations

Model I shows the traditional situation of government (G) confronting polluters (P), including firms and trade unions. Model II presents the government (G) confronting a divided private sector (those who lose, P, and those who gain, A and EI, with the new policy), and Model III shows a lowering of conflicts by reducing opposition by (P), increasing the acceptance level of the new policy instrument.

Model III can be further developed in two ways. On one hand, (A) and (EI) may consider seriously collaborating voluntarily with (P) through negotiations "a la Coase" when polluters can block tax implementation. This is so because if pollution is not reduced, new business will not develop. On the other hand, balance can be altered if environmental NGOs add their own pressure to solve the environmental problem.

In summary: (G) seeks to solve pollution through changes in environmental policy (taxes or tradable permits plus new enforcement mechanisms), responding to demands from the constituency, businesses that profit by this measure (A and EI), and environmental groups (NGOs). Against these changes we have the polluters (P) that press (G) to block the new policy or, if this is not possible, to adjust it in order to reduce their economic damage as much as possible. (P)'s resistance is feasible "to be defeated" by the political pressure from (A), (EI), and (NGOs) on (G), and/or through voluntary negotiations between (P) and (A) or (EI).

If (G) is able to manage these conflicts properly, there will be a higher probability of implementing the new policy, and after a transition period, the end situation may look as follows: (i) (P) polluting less; (ii) (EI) supplying equipment and services to (P); (iii) (A) developing its business—old or new—without the negative

impact received from (P); (iv) (G) monitoring new instruments' compliance; and (v) (NGOs) also keeping an eye on compliance through private enforcement mechanisms, press campaigns, etc.

In a socially optimal solution, polluters must internalize their environmental negative externalities. But conflicts of interest, such as those analyzed here, can preclude achievement of this kind of solution. Thus, this point justifies exploring implementation alternatives, although they could imply a departure from the optimum. Of course, this is not nice from a theoretical point of view, but not too bad for an environmental policy's effectiveness. As Baumol (1972) stated, "If we permit ourselves to be paralyzed by councils of perfection we may have still greater cause for regret."

5.2. Conflicts inside the government

In the above description of the three possible models, government was considered as a single actor, although actually it is often an aggregate with internal divisions that do not always share objectives and decisions. For instance, the government can be divided in different areas according to functions and in different levels according to geographic jurisdiction. And these divisions may show disagreement in relation to environmental policy design and implementation, frequently ending in conflicts that may jeopardize the environment. Below we discuss three types of government conflicts, common in developing countries: within a given government level, among different government levels, and among governments of equal level.

Conflicts within a given government level

The conflict between the environmental agency and the treasury is a typical case. The objective of the environmental agency is to reduce, say, pollution. For that purpose it seeks an environmental tax as an incentive measure and, if effective, emissions and also tax revenues will diminish. On the other hand, the treasury's objective is to collect taxes and sustain public revenue in conformity with public expenses, so the above-mentioned fall in the revenues is usually not welcomed. Therefore, the environmental agency prefers a high environmental tax rate, while the treasury prefers the opposite. In this sort of conflict in developing countries, because of traditional political differences, the environmental agency generally has less political weight.

On the other hand, coordinated actions between these areas are necessary, employing the comparative advantage of the environmental agency in monitoring and auditing, and of the treasury in collecting the tax. Of course, if taxes are not earmarked for environmental policy, the environmental agency will have less incentive to cooperate.

Conflicts among governments of different levels

This kind of conflict appears in federal systems with local and a central government, e.g., municipalities and provinces. As noted before, attribution delimitation

is crucial to avoid conflicts and make easier complementary actions. For instance, it has to be outlined what level establishes the type of tax and what level fixes the rate. And if both levels have an environmental policy, it is necessary to determine what aspects of the pollution or natural resources problem correspond to each level. Overlapping attributions lead to less effective enforcement. This is one of the weaknesses of Argentinean environmental policy, especially in relation to water pollution: it is very difficult to identify who has to control it and what agency is accountable for it.

Even if the overlapping attributions problem is overcome, strong cooperation between local and central agencies is essential: (i) the local agency brings its greater social, economic, and ecological knowledge of the area, which helps to better calculate the tax rate in relation to citizens' preferences, population density, assimilative capacity, and kinds of economic activities involved, and (ii) the central agency brings the technical capability and administrative structure for instrument designing and monitoring, a capacity that municipalities usually lack.

Conflicts among governments at the same level

If a polluted river crosses several municipalities, the optimal tax rate should be set in accordance with local characteristics and pollution externalities of downstream jurisdictions. This scheme avoids competition among municipalities offering environmental sanctuaries to increase local firms' relative competitiveness and attract investments. However, frequently there are legal barriers for this decentralized decision system.

The influence of environmental policies on competitiveness is a relative one: it depends on many factors such as tax rate level, emission magnitude, firm costs structure, and productive flexibility. The allegation that investment location decisions follow environmental policy differences is not supported by empirical evidence: the environment is only one (and not the dominant one) among multiple elements that firms take into account when choosing a location.

Cropper and Oates (1992) claimed that the cost of carrying out environmental regulations is not significant, and Jaffe et al. (1995) presented results of their study in the United States showing that the installation of new plants did not follow a pattern determined by differences in the rigorousness of the environmental policies. Similar conclusions for India were obtained by Mani et al. (1997). However, if the environmental policies increase their rigor, it cannot be dismissed that in the future their importance may grow in cost structures and in localized decision-making.

On the other hand, it has been argued that if each municipality establishes the tax in a decentralized way, it will not consider interjurisdictional externalities but only the local impacts. There is not a unique solution to this problem: decisions could be in the hands of the central (provincial) authority or arising from a cooperative agreement among the riverside jurisdictions. Basin Committees or Watershed Authorities should agree on the characteristics of the instrument to be applied and the time horizon for its implementation and, why not, to negotiate solutions,

with municipalities from above and below the pollution source collaborating among them (OECD, 1991). Of course, establishing a Basin Committee is much simpler than making it operative.

6. Conclusions in the Framework of the Knowledge Revolution

We have postulated that the employment of the economic instruments complemented by the newer dissuasive mechanisms of enforcement constitute one way of improving the effectiveness and efficiency of environmental policies, even in developing countries.

A change in the intensity of current uses of economic instruments implies that some sectors lose and others win. If the pressure capacity of the winners overcomes that of the losers, the political feasibility of its implementation will increase. This conflict of interests can also be solved by helping those that are harmed by the change through subsidies and technical assistance, financed from diverse sources.

Implementation also brings internal conflicts to governments, inside one jurisdictional level, among jurisdictions of different levels or among jurisdictions of the same level. Appropriate attention to the conflicts among the diverse private and government social actors is essential to improve the probability of success of changes in the environmental policies.

However, modification of environmental policies would be more effective and efficient if: (a) the cost of environmental technologies were lower, (b) it were easier to have access to "clean" technologies, (c) the environmental authority had better audit and control means, (d) citizenry pressed for better care of the environment, and (e) consumers watched more closely the environmental behavior of companies when they decide what product to buy.

Presently this change of scenario is more probable than before, because society is experiencing transformations originated in the way people use knowledge. This was called the Knowledge Revolution by Chichilnisky (1997).

The Knowledge Revolution is making society more innovative and knowledge-dependent, so that knowledge-intensive sectors will constitute a growing portion of the economy. These sectors are able to generate a series of innovations that can contribute to improvement in the care of the environment. For example:

1. *New production techniques:* the development of more knowledge-intensive production techniques will allow agricultural and industrial production to become more environmentally friendly.
2. *New auditing techniques:* advances in technologies to control the environmental performance of companies will improve the effectiveness of the audit and make easier the application of economic instruments of environmental policies.

3. *New enforcement procedures:* the emergence of private enforcement procedures opens a new set of alternatives to improve the execution of environmental policies.

Last but not least, education plays a central role in this revolution. Why? Societies with little social mobility face fewer possibilities for innovation and change; as a consequence, they can take little advantage of the Knowledge Revolution in all its aspects, including the environmental ones. Education is an excellent instrument to increase social mobility and to improve adaptability to change, helping, in turn, to promote values for caring for the environment that make more effective mechanisms as the green stamp, the private enforcement, and self-control.

The transformation that society is presently experiencing will greatly improve the effectiveness of more efficient instruments of environmental policies over those applied traditionally, collaborating in the search of a path for sustainable development.

Acknowledgments

We thank Víctor Beker, Osvaldo Baccino, and an anonymous reviewer for helpful comments and suggestions. Of course, we are responsible for all remaining errors.

Literature Cited

Banco Mundial. 1992. Informe sobre el Desarrollo Mundial 1992. Desarrollo y Medio Ambiente, BIRF, Washington, D.C.

Barde, J. P. 1994. Economic instruments in environmental policy: Lessons from the OECD experience and their relevance to developing economies. OCDE/GD93193, Paris, January.

Baumol, W. J. 1972. On taxation and the control of externalities. American Economic Review 52: 307–322.

Becker, G. 1968. Crime and punishment: An economic approach. Journal of Political Economy 76: 169–217.

Bovenberg, A. L. & L. H. Goulder. 1996. Optimal environmental taxation in the presence of other taxes. American Economic Review 86: 985–1000.

Bruce, N. & G. M. Ellis. 1993. Environmental taxes and policies for developing countries. WPS 1177, Policy Research Department, The World Bank. Washington, D.C.

Chichilnisky, G. 1997. Ecology and the Knowledge Revolution. Keynote address to the Second National Forum on Biodiversity, "Nature and Human Society." National Academy of Sciences, Washington, D.C., October 30, 1997.

Chudnovsky, D., A. López & V. Freylejer. 1997. La prevención de la contaminación en la gestión ambiental de la industria argentina. CENIT, DT 24, Octubre.

Cropper, M. L. & W. E. Oates. 1992. Environmental economics: A survey. Journal of Economic Literature 30: 675–740.

Eskeland, G. S. & E. Jimenez. 1992. Policy instruments for pollution control in developing countries. World Bank Research Observer 7: 145–169.

Galperín, C. D. 1995. Bienes públicos, free rider y externalidades: Una revisión crítica de la bibliografía para su aplicación al estudio del medio ambiente. Libertas 23: 189–242.

Goulder, L. H. 1998. Environmental policy making in a second-best setting. Journal of Applied Economics 1: 279–328.

Hamilton, J. T. 1995. Pollution as news: Media and stock market reactions to the Toxic Release Inventory data. Journal of Environmental Economics and Management 28: 98–113.

Helm, D. & D. Pearce. 1990. Assessment: Economic policy toward the environment. Oxford Review of Economic Policy 6: 1–16.

Jaffe, A. B., S. R. Peterson, P. R. Portney & R. N. Stavins. 1995. Environmental regulation and the competitiveness of U.S. manufacturing: What does the evidence tell us? Journal of Economic Literature 33: 132–163.

Konar, S. & M. A. Cohen. 1997. Information as regulation: The effect of community right to know laws on toxic emissions. Journal of Environmental Economics and Management 32: 109–124.

Laffont, J. J. & J. Tirole. 1993. A Theory of Incentives in Procurement and Regulation. MIT Press, Cambridge, Massachusetts.

Mani, M., S. Pargal & M. Huq. 1997. Does environmental regulation matter? Determinants of the location of new manufacturing plants in India in 1994. World Bank Policy Research Working Paper 1718. February. Washington, D.C.

Margulis, S. 1993. The Experiences of Brazil, Mexico, Chile, and Argentina in the Use of Economic Instruments in Environmental Policy. SG/IW/EPOC936, Background paper N° 3, presented to the Informal Workshop on the Use of Economic Instruments in Environmental Policies, OECD, Paris.

North, D.C. 1984. Estructura y Cambio en la Historia Económica. Alianza Editorial, Madrid.

Oates, W. E., P. R. Portney & A. M. McGartland. 1989. The net benefits of incentive-based regulation: A case study of environmental standard setting. American Economic Review 79: 1233–1242.

OECD. 1989. Economic Instruments for Environmental Protection. OECD, Paris.

OECD. 1991. Environmental Policy: How to Apply Economic Instruments. OECD, Paris.

OECD. 1992. The OECD Environment Industry: Situation, Prospects and Government Policies. OCDE/GD 92 1, OECD, Paris.

OECD. 1993. The Role of Economic Instruments in OECD Member Countries. SG/IW/EPOC934, Background paper N° 1, presented at the Informal Workshop on the Use of Economic Instruments in Environmental Policies, OECD, Paris.

OECD. 1996. Implementation Strategies for Environmental Taxes. OECD, Paris.

Palange, R. C. & A. Zavala. 1987. Water Pollution Control. Guidelines for Project Planning and Financing. World Bank Technical Paper Number 73, World Bank, Washington, D.C.

Palmer, K., W. E. Oates & P. R. Portney. 1995. Tightening environmental standards: The benefit-cost or the no-cost paradigm? Journal of Economic Perspectives 9: 119–132.

Pearce, D. W. & J. Warford 1993. World Without End: Economics, Environment and Sustainable Development. Oxford Univ. Press, New York.

Polinsky, M. & S. Shavell. 1979. The optimal tradeoff between the probability and magnitude of fines. American Economic Review 69: 880–891.

Porter, M. E. & C. van der Linde. 1995. Toward a new conception of the environment-competitiveness relationship. Journal of Economic Perspectives 9: 97–118.

Rabinovich, J. 2001. Environment and economics from the perspective of the South. *In* Managing Human-Dominated Ecosystems. Monogr. Syst. Bot. Missouri Bot. Gard.

Repetto, R., R. C. Dower, R. Jenkins & J. Geoghegan. 1992. Green Fees: How a Tax Shift can Work for the Environment and the Economy. World Resources Institute, Washington, D.C.

Tietenberg, T. 1996. Private Enforcement of Environmental Regulations in Latin America and the Caribbean. An Effective Instrument for Environmental Management? ENV-101, BID, Washington, D.C.

USEPA. 1990. Environmental Enforcement. A Citizen's Guide. Washington, D.C., U.S.E.P.A., March.

World Bank. 1995. Argentina. Managing Environmental Pollution: Issues and Options. Report N° 14070-Argentina. World Bank, Washington, D.C.

Environment and Economics from the Perspective of the South

Jorge Rabinovich
Departamento de Investigatión
Universidad de Belgrano
Zabala y Villanueva
(1428) Buenos Aires, Argentina

...but here below below
each one in its hiding place
there are men and women
that know what to struggle for
taking advantage of the sun
and also of the eclipses
leaving aside the useless
and using what doesn't serve
with its veteran faith
the South also exists

Mario Benedetti
Uruguayan poet

Abstract

The "South" can be characterized by seven social-political-economic "marks": (1) inequity, (2) natural resource extraction, (3) economic concentration, (4) weak/vanishing governments, (5) economic depletion, (6) legal vacuum and judicial insecurity, and (7) political decadence. The world is experiencing a strong globalization and privatization process and a tendency toward "knowledge-intensive" economies, and the South will inevitably get involved in this course. I here analyze whether under these conditions the South can assimilate this process. Private capital flows to the South at an accelerated rate and has strong international mobility. The shift from private to multilateral international loans comes with a structural adjustment "package." Both processes create conflicting objectives in governments in the South, putting pressure on environmental policy decisions. Subsidies create

market distortions that contribute to environmental deterioration. Costs from natural services can be estimated but imply a dependence on information and knowledge limited in the South. Distributional and equity issues are prevalent, with negative environmental consequences. Free trade aggravates environmental deterioration and over-exploitation because the South will not resist the pressure to extract primary commodities. Internalizing environmental goods and services, with their respective institutional and policy changes, should precede free trade decisions. Effective institutions, participation of stakeholders, and clearly defined property rights are essential for a "knowledge-intensive" economy. Countries in the South will have to chose between creating and implementing costly state institutions to protect rights and monitor production, or to keep an open access regime costly to the environment. Non-governmental organizations (NGOs) are now receiving in the South appropriate credit as legitimate organizations for environmental negotiations, although measures to improve their accountability are still lacking. Legally, public goods are defined as those belonging to the state, imposing serious restrictions toward "knowledge-intensive" economies, unless property rights of public goods can be considered. Well-specified, transferable property rights (also involving intellectual property rights) to natural resources result in incentives to collect information on alternative uses and a reallocation among those uses. Property rights, as combinations along a spectrum from open access to private ownership, are a necessary but not sufficient condition for resource sustainability: they have to be well specified, context-specific, and enforceable. The civil codes of many Latin American countries are legal barriers to these processes. Frequently, Latin American states cannot avoid environmental degradation due to their inability to carry out environmental policies, to enforce regulations, and because of a weak civil society. Nature cannot be divided by political and/or economic criteria: a notion of ecological economies of scale is proposed to replace the equivalent notion of economies of scale.

1. Introduction

During the colonialist European expansion of the last century a variety of agricultural and industrial practices, successful in their original countries, were carried to developing countries as technological packages, and transferred despite cultural and climatic differences. Later the Green Revolution, introducing high-yield, genetically uniform cereals, as well as farm equipment and other chemical adjuncts, even irrigation schemes, were transferred to developing countries, with a few outstanding successes but many more failures (Viglizzo, in press).

With a worldwide-strong globalization and privatization process, knowledge-intensive economic activities are characterized by relatively small investment and little natural resource utilization, generating moderate contamination and still maintaining a dynamic effect on the economy. This supposedly diminishes pressure on land use,

especially by reducing forest clearings, with benefits such as protection of springs and river sources (thus conserving water resources), reduction of erosion, conservation of the biodiversity, and reduction of carbon emissions to the atmosphere. Similar effects would be felt on aquatic and terrestrial non-forested ecosystems. The Knowledge Revolution (Chichilnisky, 1997a) significantly relates to property-rights aspects of physical and biological resources as well as to the intellectual property rights associated with knowledge-intensive activities (Chichilnisky, 1997b).

Can current economic, social, and political conditions in the South assimilate this tendency or will last century's European expansion with technological packages be repeated?

2. The South

The South can be broadly characterized by seven features (called marks hereafter):

(1) *Inequity*. Important social and economic differences are clearly manifested by differences in quality of life (income, education, health).
(2) *Extraction*. Natural resources are intensely extracted, generally beyond their sustainable capacity.
(3) *Economic concentration*. A tendency toward a strong concentration of economic power is perceived, often carried out by transnational companies.
(4) *Vanishing governments*. Strong privatization is taking place with inadequate control and regulatory capacity.
(5) *Economic depletion*. Economies show large fiscal deficits despite: (i) a growing external debt, (ii) a certain control of public expenditures, making more strict budget constraints, and (iii) an increase in the tax pressure on citizens and firms.
(6) *Legal vacuum and judicial insecurity*. The legal system with respect to natural resources and the environment is frequently contradictory. Property rights regimes are weak. In many of the countries of the South this is associated with corruption, and a tendency of the political power to interfere with the judicial power.
(7) *Political decadence*. The political system, despite an important democratization process in the Latin American continent, shows a political class strongly discredited by the citizenry. Outstanding characteristics are: (i) time-horizon shortsightedness with mainly an electoral goal in its political proposals, (ii) lack of idealism in the formulation of national or provincial plans, (iii) negligence and indifference in their implementation, and (iv) high levels of corruption.

3. Economy and Environment: Southern Issues

Modern technologies can improve the competitive position of countries with an economy based on natural-resource extractive activities, but that still does not

switch a country into a knowledge-intensive economy. Rates of change of ecological processes, including agricultural activities, will never attain the dynamism of electronics, communications, or finance, and so cannot compete with the knowledge-intensive industries that have become independent of the cycles of nature (Viglizzo, in press).

Although the seven marks are not exclusive to the South, their concurrency and the intensity of their mix is quite outstanding. From a Northern perspective the seven marks constitute seven *problems*; however, from a Southern perspective the seven marks reflect what the South *is*, and we should evaluate which of them should change *sine qua non* to shift progressively to a knowledge-intensive economy.

3.1. The environment in the South: globalization, adjustments, markets, and subsidies

After the debt crisis of the 1980s, a twofold process took place: (a) a massive private capital flow to the South, and (b) a shift from private to multilateral loans. In 1995 twelve countries (Argentina, Brazil, Chile, China, India, Indonesia, Malaysia, Mexico, Thailand, Hungary, Sierra Leone, and Trinidad and Tobago) received about three-quarters of all private flows to the developing world (World Bank, 1996).

These private North-South flows of capital, in the form of foreign investments, are one of the clearest evidences of the present globalization process. They have their own peculiarities: (i) an accelerated rate of growth, with $44 billion in 1990 increasing to $234 billion in 1996 (World Bank, 1996); (ii) a resolute international capital mobility, seeking the best market structure, the cheapest labor, the lowest taxes, and the most receptive countries in terms of weak environmental regulations, (iii) exporting Western "consumerism," usually aggravating the already existing gap between rich and poor (French, 1997); and (iv) despite investments in cutting-edge technologies, various types of environmental damages result, particularly with investments in the primary sector.

When private lenders were reluctant to grant credit loans, capital supplied by multilateral organizations (e.g., the World Bank and the International Monetary Fund) became critical. However, these financial resources are accompanied by economic advice leading many developing countries to structural adjustment (Dyer & Belausteguigoitia, 1996). This adjustment was typically based on: (i) nominal exchange rate devaluation; (ii) trade policy reform; (iii) removal of price distortions; and (iv) tax reform. This shifts resources toward tradable goods, thus improving the efficiency of resource allocation. However, these policies are being felt in the South as conflicting objectives, putting pressure on many environmental policy decisions (cf. Section 4).

The interconnectedness of different levels of policies, and their relationships with macroeconomic policies and environmental conservation (cf. Section 4), is well known. Repetto (1996) summarized it, for the case of forest use, as follows: forests and woodlands are threatened by demand for agricultural land, which depends on domestic demand for agricultural products, which responds to growth in

population, per capita income, and income distribution, which, in turn, depend on macroeconomic policies.

Adjustments

Many of the multilateral organizations have now extended the range of their policy-export activities into environmental conservation and sustainable development strategies. The standard strategy in this field consists of (Dyer & Belausteguigoitia, 1996): (i) integration of natural resources to markets; (ii) elimination of price distortions; (iii) definition of property rights; and (iv) integration of policies and incentive measures among sectors, and between each of these and the environment. One of the fundamental instruments to achieve integration is undistorted market prices. As we see, multilateral organizations' policies are already heading toward a basic structure more compatible with knowledge-intensive economies. However, the import of policy packages and their implementation at the national level usually dismiss the differences between regions and undermine conservation efforts, disregarding local conditions as well as non-evident links for economic sectors (Dyer & Belausteguigoitia, 1996).

Markets

As environmental goods have no market values or these are difficult to estimate, and as the markets are based on the economic valuation of goods, it is no surprise that market mechanisms do not work well for environmental goods, justifying even a minimum Government intervention (Viglizzo, in press). Incorporating wildlife species into the market system may not result in their conservation and indeed could even accelerate their extinction (Costanza et al., 1997). Species within the market system will not be conserved if their expected growth (in numbers or biomass) is less than the discount rate (Clark, 1976). Only Government controls put on species harvest would protect the resources (Clark, 1976). Clark (1976) showed that, even if the discount rate is smaller than the environmental resource's rate of growth then, in the case of an open-access exploitation of common-property resources, there will be an equilibrium in which economic rent is dissipated as pressures on the resources reach such levels that revenues exactly equal opportunity costs. But this equilibrium could occur at low levels and near the extinction risk of some populations, again demanding Government intervention.

The problem is more complex at the landscape level. In an analysis of land and land value from an ecologically based view, Gottfried et al. (1996) questioned the ability of markets to create optimal landscapes, even when traditional methods of internalizing externalities are applied. Achieving sustainable landscapes requires both sufficient ecological knowledge and institutions capable of overcoming landscape-scale market failure.

In allocating various societal capital (lands, stocks of living resources, management resources) against the generation of a flow of human-desired goods and services from the biosphere (food, clothing, housing materials) Swanson (1996) identified three

routes affecting natural resource conservation: (1) stock disinvestment; (2) management-based divestment; and (3) base resource conversions. As a consequence, there is an under-investment in relation to the environment that Swanson (1992) related to two market-related failures: the inability to appropriate (a) the value from the flow of information services from diverse resources, and (b) the value of insurance services from diverse resources.

Subsidies

Subsidies as market distortions and their relationship with environmental deterioration have recently received strong attention. Myers (1996) calculated that subsidies represent 3.8% of a global economy of $26 trillion and therefore play a prime role in the functioning of the global economy. He argued that they can be detrimental to society in that they contribute to environmental destruction and overexploitation of natural resources, e.g., agricultural subsidies leading to over-loading of croplands and pollution from synthetic fertilizers and pesticides.

In addition to being very expensive for governments, the main problems associated with subsidies are that they cause inefficiencies in production or the misuse and over-use of resources. Analyzing the case of subsidies for marine fisheries, Myers (1996) showed that despite the fact that all major marine fisheries are considered to be over-exploited and that worldwide harvests have increased almost fivefold since 1950 and catches have been declining since 1989, subsidies aimed at preserving the fishermen's jobs have aggravated this situation, generating a fishing industry with an excessive extractive capacity.

Despite obvious negative effects, subsidies are still common in the South. Lately they have been reduced as part of the adjustment policies, but many continue less obvious as special aid during situations of crisis, or through tax exemptions.

Internalizing costs from natural services

Research leading to cost estimation from natural services is very active. Pimentel et al. (1997) identified 21 different natural services based on biodiversity and put a dollar estimate on them, for the U.S. in particular as well as globally. Analyzing the cost of mismanagement and to estimate the potential loss of production can be useful. Azzoni and Isai (1994) estimated the money that the government of the state of São Paulo, Brazil, should transfer to counties in which environmentally protected areas had been established by the state. Their results showed that protecting 7.68% of the territory of the state São Paulo caused a maximum decrease in its GDP of 0.05%. Howard (1996) also demonstrated that the total opportunity cost of maintaining Uganda's protected areas amounts to an annual estimated US$110 million, exceeding the gross revenues of the protected area system by 100 times and clearly suggesting that the conversion to agro-pastoral use would be financially preferable to maintaining the *status quo*. Incentive measures are clearly necessary to make environmental conservation worthwhile in economic terms. Unsworth and Bishop (1994) proposed that the public can be compensated for past losses in en-

vironmental services through the provision of additional similar services through environmental annuities.

Reid (2001) presented an innovative mechanism to capture the value of ecosystem services. The Guanacaste Conservation Area (GCA) in Costa Rica, in addition to providing economic benefits through ecotourism, local employment, and as chemicals for the pharmaceutical and agrochemical industries, has also protected a number of ecosystem services of value, which until now have been provided freely. The total value for these services was estimated at $24,000 per year and, in lieu of that payment, an orange-growing corporation donated 1200 ha of forested land to the GCA. Although acknowledgement of the value of the environmental services was not complete as to pay cash for them, this donation is a vital first step.

One possible form of incorporating natural resources into the market system is to grant to private individuals, firms, or groups the sole rights to use particular environmental resources for a fixed time period, to benefit from the economic returns from using the resource now and in the future. A consequence of this is that consumers will have to pay a higher price, reflecting the costs of conserving the natural resources. This touches the subject of institutional arrangements, which in turn intertwines with markets and adjustment policies.

3.2. The environment, sustainability, and the Knowledge Revolution in the South

It has now been widely accepted that the standard measures of economic growth do not reflect changes in environmental quality or the changing stock of natural capital (Pearce et al., 1989), and that economic growth can be faster by not internalizing environmental costs. If a country spends on environmental conservation, it will invest less in other economic, social, or political endeavors. Environmental conservation is not a business that yields immediate profits. Budgets for environmental protection are still low, even in "greener" countries, consisting of 1–1.5% Gross National Product (GNP) in Japan, Germany, the U.K., and Holland, and less than 1% of GNP in the U.S., France, Canada, Austria, Norway, and Sweden (Viglizzo, in press). Contrast with the South is tremendous: while most Northern countries invest US$30–70/person/year, in Argentina environmental investments are in the range of US$0.1/person/year (Wolinsky, 1998).

Sustainable technologies, by their nature, are extremely dependent on information and knowledge, both critical inputs difficult to manage. Unsustainable problems are not solved by physical inputs that can be bought at the marketplace: they rather demand immaterial goods as the result of an intellectual process that incorporates information, knowledge, interpretative ability, creativity, and executive capacity (Viglizzo, in press).

Prevalent globalization and privatization processes and an emerging preoccupation with the environmental questions offer conflict for those trying to eliminate trade barriers and those trying to impose barriers based on environmental principles. Those that defend the elimination of environmental trade barriers claim that it is

almost impossible to distinguish a legitimate environmental protection from a mere commercial protection, while those that try to impose them claim that the risk of commercial protection disappears if both the production and the trade of goods reflect their real environmental costs, and not the economic profits (Viglizzo, in press). May and Segura Bonilla (1997), analyzing the trade and environment debate from the South, claimed a contradiction between promoting free markets while meeting domestic environmental objectives.

Recent econometric models (Anderson, 1992) predict net effects of expanded agricultural trade as positive for the global environment. In the context of emerging regional trading blocks in Latin America (NAFTA and Mercosur), removal of trade barriers between participating nations has led to dislocation of production with localized impacts. It is hoped that they will be resolved through reconversion of agriculture in affected subregions (May & Segura Bonilla, 1997).

If knowledge-intensive economies predominate, what happens to agroforestry and other resource-extractive activities? A country rich in natural resources, once considered an important comparative advantage, finds today this a competitive disadvantage, particularly if the country's economy continues to be dependent on those natural resources (Viglizzo, in press). Argentina attained economic splendor as the granary of the world, in the first half of the 20th century, but it is doubtful it could have done so today. Increasing costs and diminishing prices of agricultural products only worsens the loss of competitiveness of most extractive activities in relation to other emergent sectors of the economies. These economic activities are becoming increasingly unpopular for being largely incompatible with sustainable use of natural resources and environmental preservation. This does not mean that agroforestry is a declining business. The world population keeps growing and demands more food, fibers, wood, and so on. Viglizzo (in press) asked what happens to an economic activity that loses profitability but remains essential for human beings. His answer was that it becomes subsidized, transformed as public service like education, health, and security. As Viglizzo (in press) recognized, this is possible in the North, where in these countries other economic sectors produce a surplus that can be used to subsidize rural production.

So, what happens in the South? Despite the theoretical advantages of free trade, governments, both in the North and South, have been protecting their own producers from competition from foreign-produced goods (sometimes to benefit vested interests). As a consequence, some market demands remain unmet due to increased import costs, thus keeping prices higher than they would have been with unrestricted trade. This artificial enhancement of a nation's competitiveness through producers, subsidies, and trade restrictions drives a wedge between private and social costs that represents a welfare loss for society (May & Segura Bonilla, 1997). Environmentally, those subsidies and trade restrictions have led to an intensification of land use, resource depletion, water shortages, and an increase in chemical usage, frequently beyond sustainability (Otero et al., 1992). The net result of these distortions is transferred as an increased pressure on natural resources. This result,

with the additional intrinsic difficulty of dealing with unpriced values for many environmental resources, produces strong market failures. This has led economists to argue that the most critical environmental resources need to be incorporated into the market system (Hanemann, 1988; McNeely, 1988; Randall, 1988). To do so, markets have to reduce distortions. To this end, economic policies, both national and from multilateral agencies, should initiate a series of correcting measures, dependent upon institutional arrangements.

3.3. Institutional arrangements

Institutional arrangements are linked with legal and political aspects of economic development. Lee (1992) has suggested several criteria for their evaluation, concluding that there are two primary requirements for sustainability: (1) the use of ecological information in decision-making; and (2) successful control over human activities, that is, what Gottfried et al. (1996) called effective institutions. Malla (1996) added that strategies and approaches for environmental conservation measures can only be effective if they consider the perspective of diverse stakeholders. However, Bhat (1996) considered that stakeholders' participation does not suffice, and that property rights issues have to be considered. At least for products derived from genetic and biochemical resources, the establishment of intellectual property rights (IPR) is certainly necessary. Developing countries have to develop suitable institutions and policies to govern use of their open-access biological resources.

Gottfried et al. (1996) examined public and private ownership in the United States to assess how well particular institutional conditions facilitate ecological adaptation for sustainable landscapes. After disregarding a strong central planning regime, they considered that a mixed public/private option, with various incentive programs and private ownership, was one of the most promising institutional arrangements. Regional governmental or quasi-governmental institutions may use incentives to coordinate landscape management by exchanging harvesting rights among private landowners, thus assuring acceptable returns on investment and avoiding taking of private property. An alternative approach examined by Gottfried et al. (1996) is the cooperative option, encouraging voluntary coalitions or partnerships that share interest in the use of a region's land. Such groups would include farmers, timber companies, government agencies, and environmental and other not-for-profit organizations.

It is clear that the owner-state determines the ground-level investment incentives by its choice of management institutions for a particular region or resource (Swanson, 1996). For countries in the South this choice plays a critical role. These countries will have to define the creation and implementation of state institutions to protect rights and monitor production, or to keep an open-access regime that jeopardizes natural resources but costs the owner-state nothing in the form of state spending requirements.

With small variations, it is readily accepted that any new kind of institutional arrangement should be based on four pillars: the Government, the private sector,

the unions, and non-governmental organizations (NGOs). However, these organizations have historically struggled for sectoral opportunistic advantages.

Until recently, Southern NGOs were considered to be confrontative and distrustful organizations. They are now beginning to receive appropriate credit and acknowledgement as legitimate entities for environmental negotiations. However, as Eberlee (1998) pointed out, if in a democratic society governments are held publicly accountable through elections, and the private sector is held accountable through balance sheets, NGOs are relatively unaccountable for their actions and are the least accustomed to public scrutiny. Measures to improve the accountability of NGOs are needed, including how the officers are selected, whether NGOs meet their objectives, whom they represent, and where they obtain their funding. However, even if all these requirements are met by NGOs, their classical role as lobbies for sustainable development is progressively limited by globalization and mobility of capitals. French (1997: 11) stated: "... a U.S. mutual fund is buying stock in a Korean conglomerate, which is in turn investing in a lumber project in Latin America. Imagine hundreds of similar transactions, then suppose that Friends of the Earth decide to lobby investors against the lumber project: where would it intervene?"

3.4. Equity

Distributional and equity issues are crucial to sustainable development but are usually not accounted. This can be observed even with new national economic indicators such as the Sustainable National Income (the Net National Product with adjustments for the degradation of renewable and non-renewable capital) (Adger & Grohs, 1994). For the South this aspect is critical. In many countries in the South, non-sustainable development and its negative environmental consequences can be directly related to distribution and equity issues. For example, using cross-country multiple regression equations, Rock (1996) showed that a reduction of deforestation rates in poor countries depends on the degree of equality in the distribution of landholdings and the extent of rural rootlessness. This author argued that the fundamental causes of deforestation are inequalities in landholding, tenure security, and landlessness.

What is the likelihood that individuals will be willing to sacrifice now in order to conserve resources for future generations? Firey (1960, 1963) claimed that this will occur if at least two conditions are met: (1) that individuals internalize values that stress future generations, and (2) that these values prove beneficial to the individual or group yet maintain self-esteem and group identification. Some of the seven marks that characterize the South are an important curb to at least the second condition.

In conventional economic practice intergenerational equity issues are addressed by discounting costs and benefits at the social discount rates, with typical values in the order of 3 to 8% per year (Howe, 1990; Portney, 1990), although these values can be strongly questioned in the South. Independently of the discount value per

se, if costs and benefits are not discounted correctly, they may leave future generations in worse shape (i.e., not Pareto-optimal). Rabl (1996) considered this conventional approach to be erroneous in treating the time-preference component of the discount rate as if it represented creation of wealth, while it really involves only redistribution, generally limited by the duration of market transactions, especially loans. Rabl (1996) proposed a two-step discounting procedure, with the conventional social discount rate for the short term (about 30 years) and the growth rate of the economy for the long term. He claimed that the difference between the rates at which costs and benefits will grow over time is what really matters, and that for many environmental costs these rates may be close to each other during the next few generations. Implied is that some of the uncertainty in rate prediction cancels and one can treat the problem as if the effective discount rates were close to zero.

There is a huge literature on the intergenerational equity issue, as well as on the criteria and mechanisms for addressing it. An interesting example is Guatemala's Petén case, where relative stability and order after some 35 years of armed conflict resulted in a land rush to settle on uninhabited land, including the Petén, the country's final frontier. Additionally, illegal logging and oil and gas exploration are carried out inside the Reserve borders. However, despite the country's urgent need for more farmland and resources, Guatemala has chosen to set aside 19% of its land base, or 50% of existing forests, as a legacy for future generations (Conway, 1998).

4. Policies for the South

Policy is a complex business and, as Norton and Toman (1997) put it, ecology and economics still have some homework to finish. For example, economic adjustment policies promote the formation of larger, private landholdings. On the other hand, policies that intensify small-landholder agriculture can slow population movement to the agricultural frontier (Rock, 1996).

Policies on environmental issues, to become effective, require the development of enforceable joint-management institutions. For example, in the context of oceanic resources, the costliness of management failures resulted in part in the adoption of a new management institution: the 200-mile exclusive economic zone (Swanson, 1993). In this case governments chose not to utilize the open-access institution and preferred a policy of private (national) property rights.

The theme basic to all policy issues was succinctly expressed by Solow (1974). Just as natural capital is capable of generating a return as with any other investment, the state decides whether a particular resource or region is worthy of an allocation of scarce societal investment funds. This societal investment is crucial. For Swanson (1996), investing in management regimes for a particular natural resource is the state's means for inducing individual investments in that resource. Open-access regimes create incentives *not* to invest in the resources subjected to

it, for the users of the resource will not view the resources as being worthy of investment. Conversely, if the state were to institute another regime (such as a property-rights regime), then harvesters might see some individual benefit.

Under-investment has been associated with two market-related failures and also to policy-related failures. As an example, Swanson (1992) considered pertinent to agriculture: (1) policies that prevent the appropriation of value from the flow of informational services from diverse resources; (2) policies that subsidize the provision of crop insurance for specialized agricultural production; and (3) policies that generally subsidize specialized agriculture and discriminate against diverse resources. Revenue structures, trade and investment incentives, and credit, tax, and pricing incentives were also identified by Repetto (1996) as policies with important environmental effects (Galperín & Rabinovich, 2001). However, distortions due to the seven marks of the South make the road toward knowledge-intensive economies an accidental one.

The case of marine fisheries provides an excellent example of three possible policy responses to the subsidy and the overexploitation problems (Myers, 1996): (1) to use the subsidies to retrain fishermen who are put out of work through reduced catches—whether through declining stocks or policy shifts; (2) to charge foreign fishermen for the right to catch off their shores; and (3) to limit the number of tradable fishing rights to individual fishermen. Myers (1996) concluded that the last-mentioned would probably prove as the most effective.

Policies and institutional arrangements are intimately connected. The promotion of community-based groups as primary resource regulators may prove (cf. Section 5.3) the best policy to offset the foregone revenue flows from public lands and to avoid the costs of regulating private lands, particularly when public lands are too small to provide adequate revenues (Gottfried et al., 1996).

5. A Legal Framework for the South

An active globalization process with the burden of the seven marks represents a real challenge to the developing South. The switch from traditional production to knowledge-intensive economies while struggling to achieve equity and prosperity, and at the same time maintaining sovereignty, is not easy. Viglizzo (in press) believes that in the globalized world no country will be able to claim complete sovereignty over natural resources even within their borders. Commercial sanctions will continue to be the most effective dissuasion mechanism. In this tension between environmental non-tariff trade barriers and free trade, it will be the latter as the probable loser, and agriculture will be the sector most impacted.

The nature of public and private goods will probably be the center of heated discussions (Irrera et al., 2001). What is the legal definition of public goods in contrast to the economic one? For economists, public goods are those that benefit many people simultaneously (non-rivalry in consumption), and also have non-excludabil-

ity (difficult to exclude anyone from enjoying the benefits of public good resources, whether they pay for that enjoyment or not) (Colby, 1989). Thus it is no surprise that many benefits from the natural services apply to the public good, making them difficult to fully value in the marketplace (Postel & Carpenter, 1997). Rivers, lakes, and reservoirs that provide environmental and recreational benefits are now being market-evaluated despite being typical public goods.

Legally, public goods are defined as those belonging to the State. This imposes serious restrictions on any tendency to switch to knowledge-intensive economies unless property rights of public goods can be dealt with. Property rights regimes are defined as bundles of entitlements defining rights and duties in the use of natural resources, and property rules under which those rights and duties are exercised (Bromley, 1991). Hanna and Munasinghe (1995), based on contributions by Eggertsson (1990), Ostrom (1990), Bromley (1991), and Hanna (1990), listed five general principles basic to property regimes: (1) the congruence of ecosystem and governance boundaries; (2) the specification and representation of interests; (3) the matching of governance structure to ecosystem characteristics; (4) the containment of transaction costs; and (5) the establishment of monitoring, enforcement, and adaptation processes at appropriate scale. Dasgupta (1992) claimed a crucial role to property rights regimes that specify access to the natural resource base and rights of use.

Based on Gordon's (1954) mathematical analysis of the dangers of lack of ownership that leads to fisheries overexploitation, Hardin (1968) postulated the unavoidable environmental degradation with no rules for natural resources use. However, Aguilera-Klink (1994) argued that the conventional approach to the "tragedy of the commons" is mistaken, resulting from a confusion among conventional economists about common property and unrestricted access, or absence of ownership. It is presently accepted that sustaining environmental resources is not dependent on a particular property regime, but rather on a well-specified property rights regime congruent with an ecological and social context (Hanna & Munasinghe, 1995). Thus the "tragedy of the commons" (Hardin, 1968) is only one special case of property rights: the open-access (or non-property) regime. Both under private and public ownership circumstances may exist that shorten the time horizon and lead to the "tragedy of the commons" (Hanna et al., 1995), persuading these authors to claim that the three main factors that matter for environmental sustainability of resource use are: (a) that property rights are defined; (b) that they reflect the social goals for resource use; and (c) that they are enforced.

Lant (1994) argued that the delineation of property rights among wetland owners and the general public, as opposed to the maximization of net benefits, is the fundamental political issue. This therefore drives policy-making, because well-specified, transferable private property rights to natural resources give owners the incentive to seek alternative uses and to reallocate among those uses to achieve the highest value (Anderson, 1996). Insecurity of tenure generates indifference to land management and/or renewable resources use. Lack of recognition of the rela-

tionship between property and natural resources has led to many national and international failed projects with ineffectual development interventions (Barrow, 1996). Security of tenure, be it individual or communal, is of paramount importance and can only be accomplished if land and resource use is part of a development process driven by sociological and ecological issues, and not for short-term political gain (Barrow, 1996).

Even if the discount rate is smaller than the environmental resource's rate of growth, for open-access exploitation, equilibrium will be reached where economic rent is dissipated as pressures on the resources reach those levels in which revenues exactly equal opportunity costs (Clark, 1976). Those levels may frequently be near population extinction densities, thus supporting Hardin's (1968) "tragedy of the commons." However, the emphasis must be placed on the relatively *unmanaged* nature of the exploitation (Swanson, 1996). Overexploitation has often occurred historically because the harvesting activities have been poorly controlled. The best example is the case of the oceans, non-sovereign territory (beyond a 10-mile zone), with their resources available on a first come-first served basis. This installs a system of incentives based solely upon first appropriation, and implies that no individual harvester has any incentive to discontinue exploiting the resource, because anything left behind by one harvester will simply be captured by another.

The property rights issue is not restricted to physical and biological goods. It also relates to intellectual property rights (IPR). Bhat (1996) analyzed trade-related intellectual property rights (TRIPS) in terms of three types of biological resources: physically non-exhaustible resources, physically exhaustible resources under current use, and physically exhaustible potential future resources. He concluded that each of these resources may require different degrees of IPR protection for their sustained use.

Another aspect of the property rights issue was provided by Chichilnisky (1994a, 1994b), who showed that if the South has ill-defined property rights on environmental resources and trades with a region with well-defined property rights (the North) then the problem of the commons is enlarged, and the North overconsumes underpriced resource-intensive products imported from the South.

Beaumont and Walker (1996), using optimization algorithms, found that, under certain conditions with access to farming technology and off-farm employment opportunities, private property does not lead to the best environmental outcome as compared to common property and open access. Market failures frequently relate to the difficulties entrepreneurs have in establishing property rights. However, Anderson (1996) claimed these failures may not be as severe as it was assumed, as shown by the Merck-INBio contract in Costa Rica (Deacon & Murphy, 1993). Property rights evolve according to the costs and benefits of defining and enforcing property rights and contractual terms (Demsetz, 1967, 1988; Anderson & Hill, 1975). As the value of biodiversity and natural services are recognized and rise, contracts by environmental entrepreneurs attempting to capture these values will increase.

Even with well-specified property rights, environmental resource use may be inconsistent with social goals. Uncertainties caused by political upheaval, health risks, and financial and macroeconomic instability may trigger consumption levels at unsustainable resource extraction rates (Hanna et al., 1995). However, it also happens that owners think about resources not only for themselves, but also for their children and grandchildren (Young, 1993; Mäler, 1994; Norton, 1992).

In the South the property-rights issue cannot be understood independently of its history. Typically, huge stretches of land were given almost for free; institutions were frequently run for the benefit of those in charge and for their friends and relatives, and graft and profiteering were widespread (see Appendix). Spanish and Portuguese colonization of America was conceived as a source of goods rather than as a place to extend the network of production and co-development. These roots were reinforced toward insustainability by various forms of individual moral and religious attitudes and schemes. Hypocrisy, although not exclusive to the Iberian part of the European continent, was rampant; the most impressive evidence is given by the way that orders from the Spanish Crown were taken by their representatives (the "*Virreyes*") in America: "obey but do not comply" ("*que se acate pero que no se cumpla*"). Coincidentally, in a recent survey among multinational firms and institutions about their perceptions of the level of corruption in different countries (zero points most corrupt, ten points least corrupt), Spain received a score of 4.3 points, close to the Latin America average of 3.7 points (seven countries), while the average score of the South was 3.4 points (24 countries) and the average of the North was 7.4 points (26 countries) (TI, 1996).

The interesting question posed by Swanson (1996), "Why are open access regimes employed by human societies?" will have a different answer from a Northern than from a Southern perspective. In the North the problem is seen as the cost between cooperation and conflict: societies would benefit from joint management of the common resource, but each would do better individually if others engaged in restraint while they pursued exploitation unconstrained (the "prisoners' dilemma"). The South shares in part the same dilemma but exacerbated by a strong individualistic attitude, and the seven marks given in section 2. This combination requires the development of enforceable joint management institutions for its resolution. Maybe all human societies, and not only those of the South, chose not to utilize the institution of open access as its costliness became apparent (Swanson, 1996). The marine fisheries example of the 200-mile exclusive economic zone portrays a sort of private (national) property-rights response to the costliness of joint-management failures at the international level (Swanson, 1993). However, this experience cannot be simply extended to terrestrial resources, for on land nations carry the responsibility for managing the resources associated with any piece of territory (Swanson, 1996). In this case the continuing existence of open-access regimes for terrestrial resources becomes a matter of national choice for federal governments.

Science and technology are also undergoing changes: researchers are less and less economically neutral. Knowledge is valuable: those generating it charge for

it, and those needing it pay for it. Although the notion of knowledge appropriation is not new, the present extremely rapid rate of change in knowledge and technological transfer, under highly competitive conditions, makes it unthinkable that commercially valuable technologies could freely integrate public domain (Viglizzo, in press). However, not all knowledge can be appropriated, as is the case of certain agricultural technological services and processes (seeds, fertilizers, farming equipment, veterinary products, etc.) because they are, in principle, openly available to anyone since there is nothing that guarantees commercial ownership.

Hanna et al. (1995) considered the proposition that natural resources and their ecosystems could be privatized, which is based on the argument that only under sole ownership would people have incentives to protect the flow of services from resources into the future, thus eluding the "prisoners' dilemma" trap of overuse. In some contexts collective, decentralized regimes are a more appropriate structure for environmental resource management than individual private property regimes. Hanna et al. (1995) claimed that there are three well-established general principles: (a) property-rights regimes do not exist as two opposing types but rather as combinations along a spectrum from open access to private ownership; (b) property-rights regimes are necessary but not sufficient conditions for resource sustainability; (c) no single type of property-rights regime can be prescribed as a remedy for problems of resource degradation and overuse. However, both effective control and ineffective control can exist under any kind of regime.

Since the early proposals for privatization, a rapidly expanding body of research (National Academy of Sciences, 1986; McCay & Acheson, 1987; Berkes, 1989; Berkes et al., 1989; Martin, 1992; Pinkerton, 1989; Berkes & Feeny, 1990; Feeny et al., 1990; Ostrom, 1990; Bromley, 1992) has undermined the argument for private property as the single solution for resource problems. Thus it is now fair to ask whether well-specified property rights should belong to individuals and/or to firms, or to the commons?

In the communal lands of Africa, Barrow (1996) recognized two sets of contrasting tenure rules: statutory and customary, frequently co-existing. In statutory law land is treated as a single resource, while in customary law it is not. Government policy tends to promote privatization of land for farming, since land suitable for cultivation is thought to be underused if left pastoral. Communal dry-land management practice encouraged the use of protected seasonal grazing reserves, rich patches of vegetation, clan rights to water, and in some cases specific rights to trees. In those communal lands the authority of the elders has been eroded in favor of a distant national authority, so that customary management rules are increasingly difficult to enforce.

The unpredictability of natural resources has many sources (e.g., meteorological, social, or the dynamics of the ecological system; Holling, 1978) and to cope with it human societies have resorted to all sorts of political arrangements. In relation to ecological setting and pasture availability in the desert, Perevolotsky (1987) found that the Southern Sinai Bedouin tribes practice a policy of "open grazing

territories" in which neighboring tribes are permitted to exploit rich pasture within otherwise well-defined borders of another tribe's territory. Such behavior contrasts with the conventional wisdom on pastoral nomadic territoriality. Perevolotsky (1987) argued that the reason is lush spring pasture, a major requirement for livestock production, which grows in patches whose distribution and abundance are determined largely by unpredictable rainstorms. So even large territories are not sufficient to ensure year-after-year fodder for the tribesmen's herds. It is advantageous for these pastoralists to share their temporary, dense resources, since it allows them access to their neighbor's pasture once their own territory becomes barren. Perevolotsky (1987) considered that it is the loose social and political structure of this kind of nomadic pastoral societies that contributes to the development of open grazing territories.

Barrow (1996) recognized four advantages of community control over natural resources: (1) resources are managed as a whole, and their use spreads over a wider area; (2) products can be distributed more equitably across the community; (3) the community uses the natural resource base as an asset to meet community needs; and (4) the community may be better able than individuals to protect the resources against incursions from outside. Governance by local communities is more effective and efficient on the grounds that ecological conscience and identity are more likely to be a product of community life than of regional or national collectivities (Lee, 1992). This somehow reflects an old saying: "The closer to you, the closer to your heart." Costanza et al. (1997) put a strong emphasis on fostering community:

> "Community economic life can be disrupted not only by fellow citizens who, though living in another part of your country, might at least share some tenuous bonds of community with you, but by someone on the other side of the world with whom you have no community of language, history, culture, law, and so on."

However, when disinformation and a government's interagency coordination difficulties can be overcome, the use of public lands to provide ecosystem services may prove most cost-effective, for the avoided costs of regulating private lands may easily offset the foregone revenue flows from public lands (Gottfried et al., 1996). However, if public lands are too small to provide adequate revenues, either targeted policies and/or promotion of community-based groups as primary resource regulators may prove more effective.

Emerton (1996) examined the local and communal uses of forest resources by the Maasai people around Oldonyo Orok in southern Kenya. There are strong incentives to maintain forest biodiversity there. Pasture, water, and forest are governed by a common set of rules and institutions, defined and enforced by a local group of leaders drawn from age-set spokesmen and male elders. This decision-making body operates on the basis of "consensus, discussion, and sanctions" (Galaty, 1980) and regulates forest use according to local priorities and norms. The forest is technically open to all members of the Maasai, but other groups and individuals are not permitted to use or enter the forest (Emerton, 1996).

Cooperative solutions to ameliorate externalities may be Pareto superior compared with non-cooperative abatement strategies, but those solutions may not be adopted without appropriate institutional mechanisms. To achieve cooperative outcomes voluntarily, there must be methods of determining cost shares and identifying preferred solutions (Loehman & Dinar, 1994).

There are pros and cons of communal management of natural resources (CMNR). Positive aspects include:

1. A shorter feedback time for decision-making, allowing for rapid change of course and adaptive management strategy.
2. Greater transparency of decisions and actions, due to the closeness between members of the community and their leaders, and the participative nature of the communal organization.
3. As a result of (1) and (2) there is greater direct control over the natural resources accompanied by a tendency to conserve them.
4. CMNR facilitates "learning while doing," and closeness between members of the community simplifies and guarantees the transmission of traditional knowledge.

Several difficulties associated with CMNR have been recognized:

1. Technical knowledge about the management of natural resources frequently forces the CMNR to depend upon the advice and help of NGOs, at least during critical initial phases of a new program.
2. As activities and/or relationships with other communal organizations and/or state institutions increase, more regulations and command and control measures are needed, and in the process many of the above-mentioned advantages are lost.
3. Political power and decision-making become questions of conflict and generate clashes. It is known that there is a maximum optimal in size in most communal organizations, and above a certain dimension internal disharmony and strife result. The larger the communal group, the less it will further the common interest (Olson, 1965). When applied to common property resources, the larger the group the more suboptimal and unstable is the equilibrium (Ostrom, 1990; White & Runge, 1994; Ahuja, 1998).
4. Unless welfare improves, new generations tend to emigrate, limiting most of the advantages of community bonds, "learning while doing," and discontinuing the transmission of traditional knowledge.
5. CMNR may be conservative of natural resources but not necessarily economically efficient. This factor may not be critical if the main use of the natural resource is for subsistence or for barter with nearby communal organizations.

Examples of some of these failures are already evident. Ahuja (1998) found that community controls in regulating the use of common agricultural land in Côte d'Ivoire

is substantially inefficient, and that this was correlated with larger villages that tend to depart more from social optimality. Israel's communal organizations called "kibutzim," which combine principles of democratic decision-making and communal style of life with modern technologies, apparently do not manage their business affairs and their economic resources better than private owners. This is mainly because in these relatively small communities it is difficult to find enough people with managerial skills (Yoram Yom-Tov, pers. comm.). Davos (1998) claimed that sustainable voluntary cooperation of stakeholders with competing interests and priorities cannot result from "best" policy objectives derived from expert-based rational analyses, but rather from seeking to identify "correct" policies that can draw the maximum possible stakeholder support. Vargas (1998) stated that common regimes can result in the conservation of biodiversity while simultaneously providing economic returns to local populations under conditions of clear boundaries and membership, adequate use rules, monitoring, enforcement, and mechanisms for conflict resolution.

Hanna and Munasinghe (1995) claimed that, in questions relating to property rights and environmental resources, there are five areas characterized by large gaps in knowledge: (a) the design of governance systems, (b) the development of stewardship and equity in environmental management and its contribution to environmental resilience, (c) the management role of traditional ecological knowledge, (d) the mechanisms by which humans are linked to their environmental resource base, and (e) the relationship between poverty, population, and environmental degradation.

Hanna et al. (1995) proposed that the performance of a property rights regime may be measured in any one or a combination of three dimensions: economic, social, and ecological. For the economic dimension they proposed the measure of economic efficiency (the extent to which the production of the "best" economic outcome is produced by means of the least-cost combination of inputs). For the social dimension they proposed the measure based on the equity properties of the regime (reflecting social definitions of fairness in the distribution of benefits and costs). For the ecological dimension they proposed the measure of the extent to which stocks of natural capital are maintained.

Unless resources are in surplus relative to demand, tradeoffs between these performance measures are inevitable, and in making tradeoffs uncertainty dictates a precautionary approach to cover contingencies (Costanza, 1987). In human societies, uncertainty creates incentives for accelerated rates of use due to the lack of assurance that resources not used in the present will be available in the future (Gardner et al., 1990; Bromley, 1991; Runge, 1984). Such incentives create the need for an institutional framework that constrains human actions.

In African communal lands there are frequently two contrasting and co-existing tenure rules (Barrow, 1996): statutory and customary (statutory law treats land as a single resource, while customary law does not). In central Kenya loss of biodiversity results from an open-access resource notion in rangelands between the time land is

sold and its subdivision to individual small-landholders, a period that may take several years, allowing for infrastructure and environmental damage, with illegal hunting and charcoal production rapidly depleting wildlife and forest patches (Heath, 1996).

A pilot program initiated by the Kenya Wildlife Service allows for limited wildlife user rights to landowners, altering perceptions in communities living in marginal areas that now see wildlife as a suitable land use option instead of a conflicting resource. Heath (1996) suggested as a solution to the agriculture in rangelands of Kenya the granting of property rights over wildlife to the landowners, making them responsible for its management.

There are formal legal barriers to carry out such an arrangement in Latin America (Irrera et al., 2001). Until either civil codes are modified and/or formal administrative procedures approved by law, the citizenship—particularly peasants—distrusts government's gestures of granting the use of wildlife to the landowners or the land-occupants. Rabinovich et al. (1991) found that a peasant community in the high Andes of Argentina dismissed the much more profitable use of wild populations of vicuñas because of the uncertainty of the government's political decisions regarding vicuña utilization permits in the near future.

Environmental entrepreneurship seems to be a key ingredient of free-market environmentalism (Anderson, 1996), entailing two elements: vision and contracting. The former requires some individuals to be constantly scanning the horizon in search of opportunities not seen by others, and the latter requires an institutional setting in which environmental entrepreneurs can thrive. Encouraging coordination or contracting requires a legal environment that minimizes transaction costs, conditions absent today in Latin America (Irrera at al., 2001).

6. The Political Framework

The impotence of Latin America societies to face collective actions to avoid environmental degradation is not only based in the inability of the states to carry out environmental policies and to enforce regulations, but also in the weakness of their civil societies. While the political system in the North is characterized by a representative and institutionalized polyarchy (*sensu* Dahl, 1971), Latin America seems to be consolidating a different class of polyarchy, with a citizenry of low intensity and a representation crisis. This type of polyarchy has been called Delegative Democracies (O'Donnell, 1996). These democracies have inadequate capacity for structuring options for changes, for they lead to public policies of non-decisions, particularly in the environmental arena. Public policies, either by bureaucratic action or omission, result from a process of resolving conflicts and carrying out negotiations among different actors with different projects that, after achieving decisive weight, become agents of change. Three basic types of can be recognized (Przeworsky, 1994): the state, with interests in legitimacy; the managers, with interests in profitability; and the civil society, with interests in well-being (including the well-being of the environment).

In the South the state, due to structural reforms resulting in a reduction of the size of the state as a bureaucracy, is frequently ineffective in terms of the regulations for social life. Essentially, it is not solving basic problems such as the equality of the law for all the citizens. This results in a democracy of *low citizenry intensity* (certain guarantees and rights intrinsic to the citizenship become reduced), and with unequal access to the law, evident through the institutionalization of *particularism* (vested interest groups prevail in detriment of the interests of the public at large). As a result of this private power system, citizens cannot make externalities generated by private agents become internalized through public action, e.g., externality costs of harmful activities for the environment.

Delegative Democracies are usually hostile to democratic representative schemes and to the creation and reinforcement of political institutions, resulting in ineffectual control of the Executive's actions by other autonomous public organisms. Evidence of weak horizontal responsibility is found in the high levels of Government governing by decrees or edicts instead of by laws; in political corruption; and in judicial insecurity. In this type of polyarchy it is possible to find a mix of two types of institutions: (a) a highly formalized, universal, and intermittent institution: the elections; and (b) the permanent and ubiquitous institutionalization of particularism: the clientelism, where assistance is carried out for the exchange of votes. In the Delegative Democracies of Latin American the distinction between the public and the private sphere in terms of legality and behavior becomes blurred. The South also displays a severe degradation of public service. The reduction of fiscal deficit leads to a disintegration of specialized governmental agencies and to a loss of technical personnel responsible for controlling areas of public significance. Additionally, poor horizontal control by the public administration reduces the relative cost of bribes, favoring the persistence of corruption. The state and its policies have to cope with a dilemma between the goals of economic efficiency and those of political legitimacy, and in general it remains ambivalent with respect to environmental matters.

For the second type of agent, the managerial sector, public well-being and environmental matters are subordinated to the primary goal of economic profits. Additionally, this managerial sector has a direct and deep influence on the process of public policies in environmental matters.

The third type of agent, the civil society organizations (mostly NGOs), is extremely heterogeneous, ranging from neighborhood cooperatives, production cooperatives, development societies, universities, the church, to even the media. These organizations frequently cannot clearly determine their interests. Some have a strong and immediate interest related to a direct geographical vicinity, while others have no immediate interest, but rather a symbolic one.

The possibilities of each agent influencing the others toward an increase in their interests in the environmental political game are inversely proportional to the direct and primary interest that the agent has in the environmental well-being. Frequently, the civil society possesses a direct interest in an environmental problem, but its influence is secondary and indirect. On the other hand, the managerial sector has con-

siderable possibilities of influencing public policies regarding the profitability of any project, mainly due to their abundant material resources (money and information) and their possibilities of playing outside the formal institutions (i.e., their lobby strength, which is increased by low horizontal control of the Delegative Democracies).

According to the political model of strategic choice (Collier & Nordem, 1992), the possibilities of change revolve around these agents and in the resources and strategies they deploy in a given context. Transformation processes are initiated with a change in the alignment of different social actors that breaks a situation of *non-decision* with regard to a given conflict. The possibilities of a change in environmental policies will depend upon the quantity and type of actors that integrate a coalition for that change. If a certain threshold is reached, the course will be sufficiently strong as to warrant that the transition cannot be turned back. This concept of threshold or critical mass is the essence of the model of strategic choice.

7. Epilogue: The South Changes

Many difficulties can be anticipated for the application of the knowledge-intensive economies to the South, and to circumvent them it is necessary to carry out a series of both economic and political changes. For example, there are supporters and critics of free trade in relation to the environment, with apparently clear arguments to defend their judgments. However, this debate is out of focus, for there is no optimal trade policy that will counterbalance wrong environmental policies. Assuming that inadequate (unsustainable) environmental policies are and will be frequent, from a Southern perspective the central question is: *Is free trade with wrong environmental policies worse than trade protectionism with wrong environmental policies?* The free-trade-environment debate has been usually analyzed from the North assuming that the South could respond likewise to similar policies in the North. However, in the South environmental and economic policies are not always coherent, and probably internalizing the environmental goods and services costs (with their respective institutional and policy changes) should be a step prior to any decisions made between free trade and protectionism.

In my opinion, five aspects should precede, or at least be developed in parallel to, any definite change in course toward knowledge-intensive economies in the South: (a) new forms of relationships between the stakeholders, (b) new mechanisms of civil action, (c) a change in education tendencies, (d) new legal reforms, and (e) the incorporation of ecological economies of scale into economic policies.

a new social contract

New relationships among government, private sectors, and citizenship, as well as distinct structures for their action, should be envisaged. The government and private sectors already have channels and representation for decision-making, but adequate channels and appropriate mechanisms of representation for the citizenry still have to be developed. However, it is not self-evident that the citizenry would

support a knowledge-intensive economy. Many people might feel threatened by the Knowledge Revolution: not only the disenfranchised, who might gain nothing except further economic and cultural distance from the well-off, but also the middle class, whose recent gains might vanish in the face of technical demands they cannot meet without much reeducation.

mechanisms of civil action

Recognition and incentives for mechanisms of participation of various types of civic action groups (NGOs, cooperatives, civil associations, etc.) should be enhanced. Particular care has to be dedicated to carefully avoid the interference of the political parties. A combination of classic mechanisms of civil action (assemblies, public audiences) with other procedures such as "lobbying" (in the strict sense of this term, and not as an implicit bribery system as it is presently used in the South), and effective mechanisms of arbitration and mediation, should be devised. There is an apparent increase in the sensitivity of governments to pressures from grassroots, which occurs when organizations impose pressure in depth and persistence in time. Various cartels (fossil fuel, drugs, as well as military and social ones) will exert powerful pressures against a switch to knowledge-intensive economies.

education

For Viglizzo (in press), process technologies imply a different qualitative degree of technology, composed of immaterial and intangible components, resulting from a considerable added value of information and knowledge. This type of technology requires a high intellectual cost that cannot be bought in the marketplace, except when experts sell their experience and knowledge in the form of contracts. Their application implies strong previous programming, and frequent execution and follow-up, as well as a dynamic and creative permanent adjustment. For example, in the agriculture sector, they are represented by most management technologies both at the production level (crops, pastures, soils, pests, etc.) and at the administrative level (economic, finance, accounting, etc.). Thus, education is essential in developing an attitude toward a switch to knowledge-intensive economies. Education should be a national priority if the human resources of the South are expected to be available with adequate background for a true and sustainable development. However, the past generation of educational leaders is still in power, and make only lip-service support of deep changes. A new generation has to be brought up in an intellectually different and more cosmopolitan frame of mind in order to switch to knowledge-intensive economies.

However, there are some causes for hope. Some decades ago the debate was between basic and applied science; today it is focused in the public-private science dichotomy. Pressed by scarcity of resources, the government's scientific institutions now look after either selling the knowledge they have accumulated in their universities and laboratories, or establishing technological ties with commercial firms of the private sector to distribute their innovations (Viglizzo, in press).

Intellectual property rights are gaining space associated with the growing phenomenon of privatization of knowledge. Although still embryonic in developing countries, joint ventures between the academic and commercial sectors are increasing in number and in economic resources. Under this scheme researchers lose autonomy, for funds are assigned to those areas of priority for the financing company's commercial interests. If the joint venture is successful, the diffusion of the results is restricted by the company that financed the research. Currently, the state's scientific and technological system has to operate with rigorous definitions of priorities on how to assign public resources. Public scientific and technological research institutions should solve problems that are not of interest for the private sector (problems of natural resources without private protection, of rural poverty, etc.), avoiding overlap with a private sector with which it cannot compete, thereby squandering resources (problems of national security are probably the sole exception). Complementation of roles with the private sector would be the most desirable.

Sustainability and the knowledge-intensive technological processes behind it have to be genuine. This might seem commonplace and naïve, but under widespread institutionalized corruption to proceed with a *bona fide* change is not easy, particularly where temptations are high and frequent. Awareness of the consequences of individual behavior on society and on the environment needs to become established through education. As stated by Viglizzo (in press): "Society as a whole, by receiving the potential benefits of a good behavior, must develop an ethical conscience about environmental protection, and a pragmatic vision to measure the economic and commercial risk of environmental malpractice." Trust is born from a generalized decision of not deceiving others and ourselves by being indulgent toward immorality. Immorality is always bound to be a bad business even when some evidence would point to the contrary.

novel legal forms

In the South natural resource goods (and the environment in general) are frequently part of the public domain of the State, so we have to look into new judicial forms that allow their exploitation and at the same time guarantee their care. Although legal reforms are slow, the pace of changes may accelerate them, and one step in this direction is presented by Irrera et al. (2001).

from economies of scale to ecological economies of scale

The Cuban reform that followed the 1958 Revolution aimed to eliminate the *latifundia* tenure, expropriation of land owned by foreign companies, and to gain a higher standard of living for the peasantry. These reforms began by setting a ceiling of 30 *caballerías* (1 *caballería* = 13.4 ha) on individual holdings, with a maximum of 100 *caballerías* if economic operations required such a scale. Sugar plantations were converted into cooperatives to avoid their subdivision into small uneconomical units, but later in 1963 a major reorganization of state ranches took place, and they were subdivided on the basis of crop specialization into operational units of about 469 *caballerías*.

This process was not the sole result of economic scale, but of ecological scale as well. When nature cannot be divided and many people use the resource, resources will be overexploited unless the various users employ rules governing their exploitation. The conflict between the indivisibility of nature and the use of private property for environmental management is critical (Costanza et al., 1997). In Zimbabwe, landowners that were granted individual rights for the exploitation of wildlife are turning down their property fences so that their properties' size becomes optimal regarding the territorial needs and the population size of the different wildlife species subjected to exploitation (Bernardo Ortiz, IUCN, pers. comm). The conflict between indivisibility of nature and private property allotment was recognized by the Mexican government's experience in relation to peasant land tenure. The *Ejido* organization (property of the State but held individually by peasants, and land property of indigenous communities) favored non-partitioned inheritance of land and prohibitions against selling or renting land that led to the fragmentation of household landholdings into small plots (Bellon & Brush, 1994). However, as part of structural adjustment agriculture in Mexico in 1992, the *Ejido* reform was introduced to give flexibility to the land market and help define property rights, legalizing the renting and selling of *Ejido* land among peasants, and renting to private investors. It also legalized the ownership of other land by corporations, either national or foreign, thus promoting the formation of larger, private landholdings (Dyer & Belausteguigoitia, 1996).

An integrated new dimensional scale for the management of natural resources is in order: something that I would like to call *ecological economies of scale*, and that synthesizes the simultaneous constraints set by economies of scale and the conditionality imposed by natural processes based on ecological grounds; neither should be absent if the scene has to be prepared for the Knowledge Revolution. Furthermore, commercial blocks, such as NAFTA and Mercosur, add a new dimension that makes these ecological economies of scale even more necessary.

Acknowledgments

I thank the following people for providing ideas and suggestions after reading the manuscript in various degrees of development: Iván Valiela, Woods Hole Institute; Gilberto Gallopín, Swedish Environmental Institute; Ernesto Viglizzo, INTA; and Carlos Galperín, Angel Capurro, Héctor Burgos, Néstor Legnani, and Sergio Mezza from the Universidad de Belgrano.

Literature Cited

Abel, N. 1997. Mis-measurement of the productivity and sustainability of African communal rangelands: A case study and some principles from Botswana. Ecological Economics 23: 113–133.

Adger, W. N. & F. Grohs. 1994. Aggregate estimate of environmental degradation for Zimbabwe: Does sustainable national income ensure sustainability? Ecological Economics 11: 93–104.

Aguilera-Klink, F. 1994. Some notes on the misuse of classic writings in economics on the subject of common property. Ecological Economics 9: 221–228.

Ahuja, V. 1998. Land degradation, agricultural productivity and common property: Evidence from the Côte d'Ivoire. Environment and Development Economics 3: 7–34.

Anderson, K. 1992. Agricultural trade liberalisation and the environment: A global perspective. The World Economy 15: 153–171.

Anderson, T. L. 1996. Enviro-Capitalists: Why and How to Preserve Their Habitat. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Anderson, T. L. & P. J. Hill. 1975. The evolution of property rights: A study of the American West. Journal of Law and Economics 12: 163–179.

Azzoni, C. R. & J. Y. Isai. 1994. Estimating the costs of environmental protection in Brazil. Ecological Economics 11: 127–133.

Barrow, E. 1996. Who Gains? Who Loses?—Biodiversity in Savannah Systems. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Beaumont, P. M. & R. T. Walker. 1996. Land degradation and property regimes Ecological Economics 18: 55–66.

Bellon, M. & S. Brush. 1994. Keepers of maize in Chiapas, Mexico. Economic Botany 48: 196–209.

Berkes, P. (Editor). 1989. Common Property Resources: Ecology and Community-Based Sustainable Development. Belhaven Press, London.

Berkes, P. & D. Feeny. 1990. Paradigms lost: Changing views on the use of common property resources. Alternatives 17: 48–55.

Berkes, P., D. Feeny, B. J. McCay & J. M. Acheson. 1989. The benefits of the commons. Nature 340: 91–93.

Bhat, M. G. 1996. Trade-related intellectual property rights to biological resources: Socioeconomic implications for developing countries. Ecological Economics 19: 205–217.

Bromley, D. W. 1991. Environment and Economy: Property Rights and Public Policy. Basil Blackwell, Oxford.

Bromley, D. W. (Editor). 1992. Making the Commons Work: Theory, Practice, and Policy. ICS Press, San Francisco.

Chichilnisky, G. 1994a. Property rights and the dynamics of renewable resources in North-South trade. Pp. 15–54 *in* C. Carraro (editor), Trade, Innovation and the Environment. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Chichilnisky, G. 1994b. North-South trade and the global environment. The American Economic Review 84: 851–874.

Chichilnisky, G. 1997a. Ecology and the Knowledge Revolution. Keynote address to the Second National Forum on Biodiversity, "Nature and Human Society," National Academy of Sciences, Washington, D.C., 30 October 1997.

Chichilnisky, G. 1997b. Updating Property Rights for the Knowledge Revolution. John D. and Catherine T. McArthur Lecture, Program on Multilateralism, Institute for International Studies, University of California, Berkeley, 3 November 1997.

Clark, C. W. 1976. Mathematical Bioeconomics. The Optimal Management of Renewable Resources. J. Wiley & Sons, New York.

Colby, B. G. 1989. The economic value of instream flows—Can instream values compete in the market for water rights? *In:* L. J. MacDonnell, T. A. Rice & S. J. Shupe (editors), Instream Flow Protection in the West. Natural Resources Law Center, University of Colorado School of Law, Boulder, Colorado.

Collier, D. & D. Norden. 1992. Strategic choice models of political change in Latin America. Comparative Politics 24(2): 229–241.

Conway, K. 1998. Promoting Community Resource Management in the Maya Biosphere Reserve. International Development Research Centre, Ottawa, Canada. At Internet site http://www.idrc.ca/reports/.

Costanza, R. 1987. Social traps and environmental policy. BioScience 37: 407–412.

Costanza, R., J. Cumberland, H. Daly, R. Goodland & R. Norgaard. 1997. An Introduction to Ecological Economics. St. Lucie Press, Boca Raton, Florida.

Dahl, R. 1971. Polyarchy: Participation and Opposition. Yale Univ. Press, New Haven.

Dasgupta, P. 1992. An Enquiry into Well-Being and Destitution. Clarendon Press, Oxford.

Davos, C. A. 1998. Sustainable cooperation for coastal sustainability. Journal of Environmental Management 52: 379–387.

Deacon, R. T. & P. Murphy. 1993. Swapping debts-for-nature: Direct international trade in environmental services. *In:* T. L. Anderson (editor), NAFTA and the Environment. Pacific Research Institute, San Francisco.

Demsetz, H. 1967. Toward a theory of property rights. American Economic Review 57: 347–359.

Demsetz, H. 1988. The exchange and enforcement of property rights. *In:* T. Cowen (Editor), The Theory of Market Failure. A Critical Examination. George Mason Univ. Press, Fairfax, Virginia.

Dyer, G. & J. C. Belausteguigoitia. 1996. Structural Adjustment, Market and Policy Failures: The Case of Maize. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Eberlee, J. 1998. NGOs in the "New Fractured World Order." International Development Research Centre, Ottawa, Canada. At Internet site http://www.idrc.ca/reports.

Eggertsson, T. 1990. Economic Behavior and Institutions. Cambridge Univ. Press, Cambridge.

Emerton, L. 1996. Local Livelihoods and Forest Biodiversity Loss: A Case from Kenya. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme

and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Feeny, D., F. Berkes & B. J. McCay. 1990. The Tragedy of the Commons twenty-two years later. Human Ecology 18: 1–19.

Firey, W. 1960. Man, Mind and Land: A Theory of Resource Use. Free Press, Glencoe, Illinois.

Firey, W. 1963. Conditions for the realization of values remote in time. *In:* E. A. Tiryakian (editor), Sociological Theory, Values and Sociocultural Change. Essays in Honor of Pitirim A. Sorokin. Free Press, Glencoe, Illinois.

French, H. F. 1997. When foreign investors pay for development. World Watch, May–June: 8–17.

Galaty, J. 1980. The Maasai group ranch: Politics and development in an African pastoral society. *In:* J. Galaty & P. Salzman (editors), When Nomads Settle Processes of Sedenterization as Adaptation and Response. Praeger, New York.

Galperín, C. & J. E. Rabinovich. 2001. Economics and environment in developing countries: Non-traditional policy instruments. *In:* Managing Human-Dominated Ecosystems. Monogr. Syst. Bot. Missouri Bot. Gard.

Gardner, R., E. Ostrom & J. Walker. 1990. The nature of common pool resource problems. Rationality and Society 2: 335–358.

Gordon, H. S. 1954. The economic theory of a common property resource. Journal of Political Economy 62: 124–142.

Gottfried, R., D. Wear & R. Lee. 1996. Institutional solutions to market failure on the landscape scale. Ecological Economics 18: 133–140.

Hanemann, M. 1988. Economics and the preservation of biodiversity. Pp. 193–199 *in* E. O. Wilson (editor), Biodiversity. National Academy Press, Washington, D.C.

Hanna, S. 1990. The eighteenth century English commons: A model for ocean management. Journal of Ocean and Shoreline Management 14: 155–172.

Hanna, S. & M. Munasinghe (Editors). 1995. Property Rights and the Environment. Social and Ecological Issues. The Beijer International Institute of Ecological Economics and The World Bank, Washington, D.C.

Hanna, S., C. Folke & K.-G. Mäler. 1995. Property rights and environmental resources. *In:* S. Hanna & M. Munasinghe (editors), Property Rights and the Environment. Social and Ecological Issues. The Beijer International Institute of Ecological Economics and The World Bank, Washington, D.C.

Hardin, G. 1968. The tragedy of the commons. Science 162: 1243–1248.

Heath, B. 1996. Agriculture in Rangelands: A Case Study from the Laikipia District in Central Kenya. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Holling, C. S. 1978. Adaptive Environmental Assessment and Management. International Series in Applied System Analysis. J. Wiley & Sons, Chichester, U.K.

Howard, P. 1996. The Opportunity Costs of Protected Areas in Uganda. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World

Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Howe, C. W. 1990. The social discount rate. Introduction to special issue on social discount rate. Journal of Environmental Economics and Management 18: S-1.

Irrera, F., M. Oneto & J. E. Rabinovich. 2001. Property rights of public natural resources: A new legal alternative for developing countries. *In:* Managing Human-Dominated Ecosystems. Monogr. Syst. Bot. Missouri Bot. Gard.

Lant, C. L. 1994. The role of property rights in economic research on U.S. wetlands policy. Ecological Economics 11: 27–33.

Lee, R. G. 1992. Ecologically effective social organization as a requirement for sustainable watershed ecosystems. *In:* R. J. Naiman (editor),Watershed Management Balancing Sustainability and Environmental Change. Springer-Verlag,New York.

Loehman, E. & A. Dinar. 1994. Cooperative solution of local externality problems: A case of mechanism design applied to irrigation. Journal of Environmental Economics and Management 26: 235–256.

Malla, Y. 1996. A Case Study of Stakeholders' Conflicting Interests and Actions. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland. (At Internet site http://economics.iucn.org/98-03-28.htm.)

Mäler, K.-G. 1994. Economic growth and the environment. *In* L. Pasinetti & R. Solow (editors), Economic Growth and the Structure of Long-Term Development. MacMillan, London.

Martin, F. 1992. Common Pool Resources and Collective Action: A Bibliography, Volume 2. Workshop in Political Theory and Policy Analysis. Indiana University, Bloomington, Indiana.

May, P. H. & O. Segura Bonilla. 1997. The environmental effects of agricultural trade liberalization in Latin America: An interpretation. Ecological Economics 22: 5–18.

McCay, B. J. & J. M. Acheson (Editors). 1987. The Question of the Commons: The Culture and Ecology of Communal Resources. Univ. Arizona Press, Tucson.

McNeely, J. A. 1988. Economics and Biological Diversity: Developing and Using Economic Incentives to Conserve Biological Resources. International Union for the Conservation of Nature (IUCN), Gland, Switzerland.

Myers, N. 1996. Perverse subsidies. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland.(At Internet site http://economics.iucn.org/98-03-28.htm.)

National Academy of Sciences. 1986. Proceedings of the Conference on Common Property Resource Management. National Academy Press, Washington, D.C.

Norton, B. G. 1992. A new paradigm for environmental management. *In:* R. Costanza, B. G. Norton & B. D. Haskell (editors), Ecosystem Health: New Goals for Ecosystem Management. Island Press, Washington, D.C.

Norton, B. G. & M. A. Toman. 1997. Sustainability: Ecological and economic perspectives. Land Management 73: 553–568.

O' Donnell, G. 1996. Otra Institucionalización. Agora 5: 5–28.

Olson, M. 1965. The Logic of Collective Action: Public Goods and the Theory of Groups. Harvard Univ. Press, Cambridge, Massachusetts.

Ostrom, E. 1990. Governing the Commons: The Evolution of Institutions for Collective Action. Cambridge Univ. Press, Cambridge.

Otero, M., G. Estefanell & E. Trigo. 1992. Conservación de los recursos naturales, medio ambiente y comercio internacional: Una visión desde América Latina y el Caribe. IICA, San José.

Pearce, D., A. Markandya & E. B. Barbier. 1989. Blueprint for a Green Economy. Earthscan, London.

Perevolotsky, A. 1987. Territoriality and resource sharing among the Bedouin of southern Sinai: A socio-ecological interpretation. Journal of Arid Environment 13: 153–161.

Pimentel, D., C. Wilson, C. McCullum, R. Huang, P. Dwen, J. Flack, Q. Tran, T. Saltman & C. Cliff. 1997. Economic and environmental benefits of biodiversity. BioScience 47: 747–757.

Pinkerton, E. (Editor). 1989. Cooperative Management of Local Fisheries: New Directions for Improved Management and Community Development. Univ. British Columbia Press, Vancouver.

Portney, P. R.1990. Comments of discounting session. Journal of Environmental Economics and Management 18: S-63.

Postel, S. & S. Carpenter. 1997. Freshwater ecosystem services. *In:* G. C. Daily (editor). Nature's Services. Societal Dependence on Natural Ecosystems. Island Press. Washington, D.C.

Przeworsky, A. 1994. Algunos problemas en el estudio de la transición a la democracia. Chapter 2 *in:* S. O'Donnel & G. Whitehead (editors), Transiciones desde un Gobierno Autoritario. Perspectivas Comparadas. Paidós, Barcelona.

Rabinovich, J. E., A. F. Capurro & L. L. Pessina. 1991. Vicuña use and the bioeconomics of an Andean peasant community in Catamarca, Argentina. *In:* J. G. Robinson & K. H. Redford (editors), Neotropical Wildlife Use and Conservation. Univ. Chicago Press, Chicago.

Rabl, A. 1996. Discounting of long-term costs: What would future generations prefer us to do? Ecological Economics 17: 137–145.

Randall, A. 1988. What mainstream economists have to say about the value of biodiversity. Pp. 217–223 *in* E. O. Wilson (editor), Biodiversity. National Academy Press, Washington, D.C.

Reid, W. V. 2001. Capturing the value of ecosystem services to protect biodiversity. *In:* Managing Human-Dominated Ecosystems. Monogr. Syst. Bot. Missouri Bot. Gard.

Repetto, R. 1996. Macroeconomic Policies and Biodiversity Loss. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland.(At Internet site http://economics.iucn.org/98-03-28.htm.)

Rock, M. T. 1996. The stork, the plow, rural social structure and tropical deforestation in poor countries? Ecological Economics 18: 113–131.

Runge, C. F. 1984. Institutions and the free rider: The assurance problem in collective action. Journal of Politics 46: 154–181.

Solow, R. 1974. The economics of resources or the resources of economics. American Economic Review 64: 1–14.

Swanson, T. 1992. Economics of a biodiversity convention. Ambio 21: 250–257.

Swanson, T. 1993. The International Regulation of Extinction. MacMillan, London.

Swanson, T. 1996. The Underlying Causes of Biodiversity Decline: An Economic Analysis. Paper presented at the Workshop "Economics of Biodiversity Loss," organized by the Biodiversity Policy Coordination Division of IUCN-The World Conservation Union, and in collaboration with UNEP-United Nations Environment Programme and WRI-World Resources Institute. 22–24 April 1996, Gland, Switzerland.(At Internet site http://economics.iucn.org/98-03-28.htm.)

TI (Transparency International). 1996. Newsletter, Sept.: 5–7.

Unsworth, R. E. & R. C. Bishop. 1994. Assessing natural resource damages using environmental annuities. Ecological Economics 11: 35–41.

Vargas, A. M. 1998. Tenure in the context of sustainable use in Latin America. Its relationship with accountability in resource use and benefits sharing. Pp. 19–22 *in* IUCN, The Bratislava Papers on Tenure. IUCN, Gland, Switzerland.

Viglizzo, E. F. In press. La Trampa de Malthus. Agricultura, Competitividad y Ambiente en el Siglo 21. Editorial El Ateneo. Buenos Aires.

White, T. A. & C. F. Runge. 1994. Common property and collective action: Lessons from cooperative watershed management in Haiti. Economic Development and Cultural Change 43: 1–42.

Wolinsky, J. 1998. Gastos Ambientales. Lecture Notes, Universidad Argentina de la Empresa (UADE), Buenos Aires.

World Bank. 1996. Trends in Developing Economies. Washington, D.C.

Young, O. 1993. The "New Institutionalism" in International Relations. Beijer Discussion Paper Series Number 43, Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences, Box 50005, 104 05 Stockholm, Sweden.

Appendix. Case Studies of the Southern Perspective.

Land is not only an important factor of production but also a store of value and wealth, as well as a status symbol and source of social and political influence; thus it is no surprise that it has been the motive of struggles throughout human history as well as the target of a variety of social experiments in relation to land tenure. This is so because the laws governing land tenure establish land ownership, which in turn determines patterns of wealth and income distribution, which in turn bring about social and political influence. One of the oldest historically recorded land reforms was carried out in Greece in the 6th century B.C., and a Roman land reform law (*lex agraria*) was passed by Tiberius and Gaius Gracchus between 133 and 121 B.C., with popular support and serious resistance by the nobility. Since then countless land reforms took place in modern times in all continents, their basic objectives being a change in land tenure regimes, that is, a transformation in the pattern of distribution of rights to the land and of the benefits *and responsibilities* derived from its ownership. However, in more recent times land reform, in re-

sponse to systems of national planning, has changed to an integrated and complex system that, in addition to the plain land tenure change, involves a variety of activities such as patterns of cultivation, scale of operations, terms of tenancy, rural credit, marketing, state of technology, and education.

Agrarian land reforms took place in many Latin American countries, as early as 1910 in Mexico, and as recently as 1969 in Peru. Prior to these agrarian reforms most of the land, agricultural and non-agricultural alike, was under the ownership of large individual ranches ("haciendas") or peasant communities. With the agrarian reform arable land was transferred to some sort of agricultural cooperative. These cooperatives have replaced the "haciendas," and the peasants who worked in the old states on a permanent basis were given communal ownership of the tillable land. The cooperatives have, in general, some sort of management carried out by their members, usually with some kind of political decision-making institution like general assemblies and managerial committees. Important decision-making questions relate to problems such as which fields will be grazed and when, how the benefits from the resources used will be distributed, the use of stubble fields, and many more.

Below I present two case studies that exemplify several problems in Latin American countries related to property rights and natural resources conservation.

1. Patagonia steppe ecosystem management

1.1. Introduction

Patagonia is located between latitudes 36° and 56°S, and covers a surface of about 800,000 km^2, involving five Argentinean provinces: Neuquén, Río Negro, Chubut, Santa Cruz, and Tierra del Fuego. Since the end of last century most of this territory has been subjected to intense use, mainly by sheep ranches, and historians consider that colonization of Patagonia by the white man was the result of the development of ranches ("estancias") for wool production (Barbería, 1996).

1.2. Climate

Despite a variety of climatic regimes and several vegetation types, Patagonia can be considered to be predominantly arid. This characteristic is caused by the "rain shadow" effect produced by the Andes, which acts as a barrier to the prevailing humid winds from the Pacific Ocean. The decrease in rainfall is extremely steep: from about 3000 mm/year in the mountains (with typical temperate forest vegetation) to about 150–300 mm/year in the central part of the continent, where the characteristic low vegetation of the steppe predominates. The analysis of this case study is restricted solely to this part of Patagonia.

As in most arid regimes, rainfall shows both important inter- and intra-annual variability, which determines the dominant pattern of vegetation distribution. From the productivity point of view, it is the inter-annual variability that characterizes

this area, mainly because grazing stocks cannot be readily adjusted to the changing availability of forage, thus creating a strong potential for soil erosion.

1.3. Vegetation

Based on bioforms, the vegetation of the Patagonia steppe falls into four physiognomic units: (a) tall shrublands, with shrubs over 1 m, (b) shrublands, with shrubs less than 1 m, (c) dwarf shrublands, plants with perennial buds close to the ground, and (d) short grasslands and meadows (Ares et al., 1990). The general aspect is characterized by scattered low bushes, generally less than 1 m tall, mixed with bunchgrasses and considerable bare ground (Redford & Eisenberg, 1992). The most common plant species in the graminous steppes is the "sweet coirón" (*Festuca pallescens*), and in the steppes of shrublands are the "neneo" (*Mulinum spinosum*) and the "bitter coirón" (*Stipa speciosa*). These three species are important for sheep management.

1.4. Wildlife

There is a variety of mammal and bird fauna in the Patagonian steppe: guanacos, the patagonian cavy, culpeo and grey foxes, mountain lions and wild cats, ostriches (rheas), geese, etc. Many exotic species have been introduced in this region: the European hare, the European rabbit, the European red deer, the Asiatic spotted deer, the European wild swine, minks, beavers, and muskrats; these species have prospered to such a degree that some have become a valuable resource, although others have become pests.

1.4.1. The guanaco

The guanaco (*Lama guanicoe*) belongs to one of the four species of South American camelids, and is an herbivore that feeds on grasses and other herbs with a strong overlap with the sheep's diet. The sustained growth of the sheep population since 1880 had a strong impact on the wild populations of guanacos. Before the introduction of sheep, the highest densities of guanacos were found in the western more humid grasslands, but presently they are more common in the drier areas, using some of the shrubs not consumed by sheep, and near the hills that offer protection from humans. Guanacos are normally killed because they are thought to be in competition with livestock and sheep, but guanaco yearlings (guanaquitos) have also been hunted for their fur (86,324 skins were exported in 1979), although nowadays commercial pressure is minimal because of an extremely low demand for guanaquito furs. From the 3 million guanacos estimated to have inhabited Patagonia at the beginning of this century, only about 500,000 remain.

1.4.2. Foxes

There are two species of foxes in the Patagonian steppe: the culpeo fox or "zorro colorado" (*Dusicyon culpaeus*) and the grey fox or "zorro gris" (*Dusicyon griseus*).

The introduction of sheep and hares in Patagonia resulted in a strong increase in the fox populations. The culpeo fox even expanded its geographical distribution. However, this increase was in the end a disadvantage: they were hunted for their fur, and one of them (the culpeo fox) became considered as a pest because it occasionally fed on lambs. Presently $25 U.S./animal is being paid by the government in an effort to reduce culpeo fox populations (López, 1998).

1.4.3. Hares and rabbits

The patagonian cavy (*Dolichotis patagonum*) or mara (also called the "creole" hare) can weigh up to 16 kg. It is a species characteristic of the most arid areas of Patagonia. It was extremely common at the end of last century (Darwin described their abundance in the *Voyage of the Beagle*) but they are rarely seen today. The European hare (*Lepus capensis*) was introduced in Argentina between 1883 and 1897 (Grigera & Rapoport, 1983), and now occurs in most of the country. It is at the same time a resource (with a harvest ranging between 5 and 10 million a year) and a pest, for it seriously competes with cattle for grasses. The European rabbit (*Oryctolagus cuniculus*) was introduced into the islands of the Beagle Channel around 1880, and by 1953 their numbers in Chile were estimated to exceed 30 million (Jaksic & Yáñez, 1983). This species appeared on the Argentinean border between 1945 and 1959 (Howard & Amaya, 1975).

1.4.4. Felines

The mountain lion or puma (*Puma concolor*) is one of the big South American felines, and it extends its range through most of Patagonia. As most of its natural prey have been extremely reduced in numbers, it feeds frequently on lambs. Smaller felines such as the Geoffroys' cat or "gato montés" (*Felis geoffroyi*) and the "gato de pajonal" (*Felis colocolo*), once very common in Patagonia, are now extremely reduced in numbers, mainly because of the fur trade.

1.5. Management

The area is mainly dedicated to sheep (approximately 70% of the animal stock) and bovine cattle raising, with natural vegetation being almost the only source of forage for both. Current practice consists basically of taking care of sanitation and reproduction; management of the vegetation is practically nonexistent (Soriano & Paruelo, 1990). In general those in charge of exploitation perceive the role of natural forage in a contemptuous and ambiguous way. The character of the processes triggered by shepherding, often gradual and subtle, make understandable this lack of perception. Being "natural," grasses are part of what apparently does not have a cost and seems not to require any care. Seasonal variations in sheep stocking rate and the eventual recovery of paddocks (a practice associated with cattle transfers between winters and summers) actually respond to restrictions imposed by snow, cold weather, or lack of water rather than to a response to supply and demand of forage or an attitude toward a rational use of the pasture.

1.5.1. Management system

The herds of sheep are assembled in the fall, and assigned to "winter paddocks," located where the rigors of the winter are less. In these paddocks the animals give birth, remaining there until December, the shearing period. Then some sheep with lambs are sent to "summer" paddocks (Soriano & Paruelo, 1990).

Stocking rate varies according to the type of paddock and the time of year. The average annual stocking rate is of the order of 0.2–0.4 sheep/ha, with extreme upper values of the order of 1.7 sheep/ha. The maximum instantaneous stocking rates take place in the summer paddocks, between January and March, and are of the order of 2.5 sheep/ha (Soriano & Paruelo, 1990; Barbería, 1996).

1.5.2. Overgrazing

Paddocks, independent of their stocking rate, in general show that (Soriano & Paruelo, 1990): (1) half of the surface is used uninterruptedly, and (2) paddock occupation is at its maximum (about 90% of its capacity) during the period of the most active vegetation growth (September to December). Additionally, most of the year bovines occupy the same paddocks as sheep. Their foraging places, however, concentrate almost exclusively on the "mallines" (humid areas of the prairies of variable size, associated with water courses or springs), except for short periods in winter.

Based on a series of indicators of the magnitude of the transformations exhibited by the vegetation, the main problems detected in sheep ranches are (Soriano & Paruelo, 1990):

(1) A decrease in the ground cover of the best foraging grass species: "coirón poa" (*Poa ligularis*), "cebadilla patagónica" (*Bromus pictus*), and the "patagonian barley" (*Hordeum comosum*) are found only in places of difficult access for the animals (in the center or nucleus of bushes). Even there plants are, in general, small and lack vigor.
(2) Absence of plantules and young individuals, both of perennial grasses and of palatable bushes.
(3) Consumption heterogeneity among the most intensively used plants: for a given grass species, a mixture of plants generally small and of reduced vigor, in the form of "rosettes" (the center is bare soil), and larger plants but with massive accumulation of dead material in the center is found. The latter are not selected by sheep because the dry material is of low foraging quality. Animals keep eating the fresh re-growth of the small plants, hindering the accumulation of reserves and so leading to their early senescens.
(4) Serious water and wind erosion. Both processes are developed simultaneously, multiplying their effects. Water erosion prevails in the area with bigger precipitation and slopes, and finer soil texture. Laminar erosion, small landslides, furrows in denuded hillsides generated by the passage of sheep, and large trenches and gullies, most of them active, cross and dry up the "mallines." The almost

continuous bovine high stocking rate in the "mallines" accelerates this process, due to trampling and the continuous removal of plant biomass. Reticular soil erosion among individuals of "coirón dulce" leaves plants placed in a sort of pedestal making them more susceptible to wind. This causes a variety of erosion forms, depending on texture and the degree of soil protection. It is common to observe widespread deflation, evident by the accumulation of soil windward of plants and by the increase of "peladales" (areas with very low plant cover). Large fires, combined with the foraging and trampling by sheep and cattle, hinder the recovery of plant cover and cause serious soil denudation.

Most of these symptoms are the result of the kind of land use: the continuous foraging and the high sheep stocking rate at the most active time of plant growth accentuates selectivity by sheep, and this determines a differential pressure on different plant species, resulting in a decrease of the grass cover of the most palatable species.

The decrease of the plant cover, the flat topography, and the low canopy are factors that, combined with frequent strong winds prevailing most of the year, result in a potentially erosive condition.

1.6. Land tenure factors

A question becomes immediately obvious: If ranches in Patagonia are in the hands of private owners, why is there evidence of mismanagement due to overgrazing of pastures? The answer can be found in the history of Patagonian colonization. I will describe the situation of the Province of Santa Cruz, which has been thoroughly studied, as it can be considered quite representative of the rest of non-mountainous Patagonia.

1.6.1. Land distribution

The legal system adopted for land distribution in the Province of Santa Cruz rewarded capital and not labor. This situation was a reflection of the government's conception that to generate a wealthy country it was necessary to put lands into production, and this could be achieved by a quick and convenient land transfer to those individuals (or firms) that were in a position of making them produce. The requisite of available capital was evident, since access to land was simple for those that could bear the installation costs and agents' services. The function of the state was limited to grant to the residents security and facilities: security because property meant a territory inhabited only by the "tehuelche" tribes and far from the occupied areas; facilities because the concession of important stretches of land were given, at low prices and by simplified procedures. These features characterized the beginning of the occupation of Patagonia, toward 1890 (Barbería, 1996).

This period also corresponded with another feature of the distribution of public lands: disorder. Lack of knowledge of the quality of the lands to be transferred, and the absence of follow-up procedures and control after lands were transferred,

resulted in a vast concentration of land in the hands of a few owners, and to a certain degree of speculation. Despite the fact that latifundia (large estates) were detected on the best land, nothing was done about it, and changes in legislation were introduced only around 1904: the maximum area granted per person or society was reduced, access to property was notably decreased, and leasing prices and investment conditions increased. Although the law now considers *ex-ante* and *ex-post* state control to the delivery of lands, in practice it was never carried out, so that the process of land concentration in a few hands continued. Everything was favorable to people with large capital, and in this way these people became owners of important tracts of land.

Since 1911, after the presidency of Saenz Peña, a new investigation was initiated, involving the revision both of the procedures and of the system of land inspection. Despite the results showing a vast concentration of land in a few hands and firms, resulting from an efficient mechanism with the participation of middlemen and even of state officials, the situation did not change.

Five years later, during the presidency of Yrigoyen (1916), a third investigation was carried out, but after what happened in 1911, there was a general misgiving. The distrust turned out to be sound: once the concentrations and the use of irregular procedures were confirmed, a decision was made to approve an ordinance establishing the expiration of many concessions; this never materialized, however.

Negative consequences befell the inhabitants that took residence after the investigations. Since the new legislation annulled the right to acquire half of the leased surface, and because of the distrust generated by the previous acts in land delivery, procedures became long and complicated. Under such a situation interested residents had only one option: to occupy lands *de facto*, thus being exposed to insecurity, to evictions, and to conflicts in the use of the land.

The steps to obtain lands became more and more difficult, since these had to be negotiated in Buenos Aires, the nation's capital. The figure of the agents kept growing until they became key for procuring lands, for they knew the characteristics of the requested lots, and could ensemble the latifundia, including the figureheads or straw men.

As a result of these processes, even at the end of this period of "colonization," the access to land in the Prov. of Santa Cruz depends on having a minimum of capital. The transfer of small properties, contemplated in the law, did not prosper since (a) the interested people did not have the necessary technical and economic support to begin producing, and (b) the area granted (a maximum of 2500 ha per person) was too small for lands with stocking rates as low as 0.2 and 0.25 sheep per hectare, many of them being also far from the ports.

From Barbería's (1996) research it becomes clear that members of the state apparatus and even agents of the provincial police, were involved in this process. In summary:

> State policies were limited to offer security to managers, adopting a legal system of land distribution to favor latifundia, in territories populated by the na-

tive tribes, and very far from the capital, and with a high ignorance of the quality and possibilities of the land.

The abundance, easy access, and low prices of land, and the weak state control after lands were transferred, in combination with the managers' objective of producing exclusively wool, generated the present character of extensive regional herding activity and the monopolization of lands, facilitated by capital availability and bonds with members of the state apparatus.

In the Province of Santa Cruz the best lands were characterized by the lowest prices and negligible conditions, better possibilities of access to the property, large tracts of land, meager investments, and easy selection of the sites, and the inverse conditions for those lands of smaller size and in the poorest locations in regard to communications.

1.6.2. Present situation

As a result of the historical process described above, when we look at today's land tenure situation, only 15.9% of the ranchers are really owners, and the rest either rent or simply "occupy" the land (Table 1).

Table 1. Types of land tenure in Patagonia (Province of Santa Cruz, Argentina).

Land tenure	Surface		Ranches	
	Hectares	Percent	Number	Percent
Owners	3,526,761	28.9	106	15.9
Tenants	6,469,041	53.0	359	53.5
Occupants	2,215,357	18.1	169	25.2
> than one type			36	5.4
Totals	12,211,159	100.0	670	100.0

It is accepted knowledge in Patagonia that about 8000 sheep are the minimum stocking rate to be able to make a good profit from sheep rearing (mainly for wool). Variations to that figure can range from 5–6000 to 10,000 sheep, depending on the condition of the pastures, climate, and the ranch's organization (a family management of the ranch would allow a much lower stocking rate than a ranch that hires laborers). In the province of Santa Cruz, in terms of the surface used for sheep production, about 8,233,500 ha (72.6%) are below the threshold of 8000 sheep, that is, below a profitable business. In terms of the number of ranches, about 94.5% of all sheep ranches have less than 8000 sheep, while only 5.1% have sheep stocks above the minimum for a profitable business (Fig. 1).

As a result of the evolution of land tenure in Patagonia, the present situation is that of an overgrazed condition showing an accelerated rate of erosion. Closer analysis shows that very few of the sheep producers are owners (about 16%), so the remaining producers have little incentive to conserve their lands in good condition, and even if they would like to, they do not have enough capital for good management of their ranches.

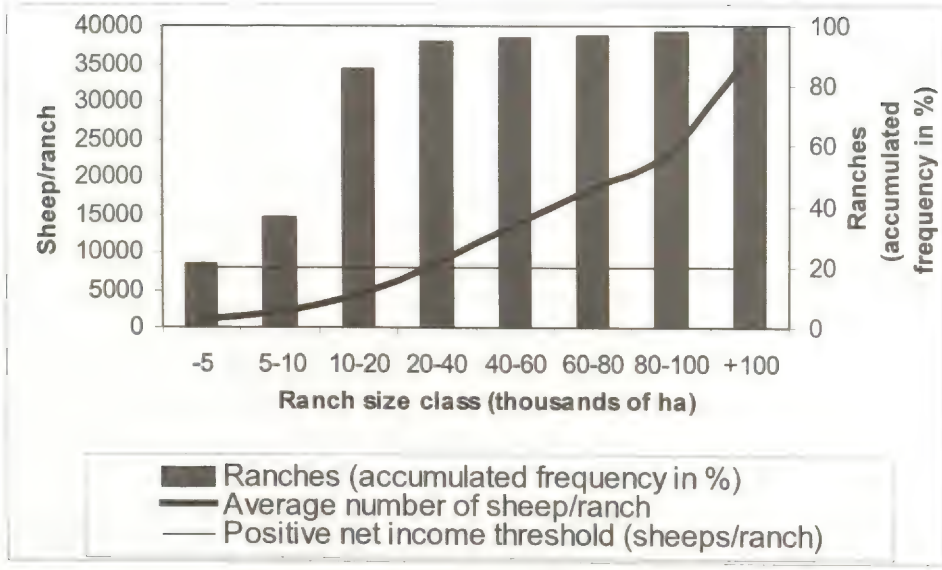


Figure 1. Cumulative distribution of the number of sheep ranches in the Patagonian province of Santa Cruz, and the number of sheep per ranch per ranch size class. The thin horizontal line represents the average sheep load per ranch that results in positive net income.

1.7. Solutions

Some of the possible solutions to remediate the overgrazed and eroded present condition of many parts of Patagonia rest on two types of complementary efforts: better management of sheep stocks, and new additional ways of using the natural resources.

1.7.1. Improved management

As indicated by Ares et al. (1990) it seems probable that, in most ranches, the recuperation of range condition might not be reached unless an overall reduction of stocking rate is attained. In some cases, redistribution of grazing pressure within a property could probably alleviate the impact that is regularly exerted upon sensitive sites. Ares et al. (1990) argued that there is a host of range management techniques which could be explored to promote redistribution of the herds in the field (e.g., the development of new forage sources in irrigated valleys and the collection of run-off water for subsequent utilization in years of scarcity). Additionally, spatial redistribution of the herd could also be attained through temporary fencing, afforestation with xerophytic species to create sheltered areas, artificial ambulatory sheltering, utilization of reserves of forage produced at sites or at times of excess, diurnal controlled herding, and more.

Of course these techniques should be applied as devised on a ranch-by-ranch basis, and a balance sheet of forage production and consumption should be built. There are government credits to those ranchers who require it for the implementation of

techniques to prevent soil erosion, but no doubt the use of these management techniques is not straightforward, and demands a knowledge-intensive technology not easily available to, or too expensive for, small landowners (Ares et al., 1990).

1.7.2. New ways of using natural resources

We have seen that productivity of this region is relatively low, but that a series of wildlife species, adapted to these conditions, prosper adequately, unless mismanaged. These species offer alternatives to local sheep producers: foxes, guanacos, hares, and felines can become a source of income from commercial activities as well as from ecotourism. Some of these species are protected, but an adequate management plan would allow for their exploitation. The success of this alternative, as expressed in the body of this chapter, depends essentially on satisfactory property rights through concessions, or similar legal figures, that assure sheep producers that they can plan ahead, but also establish their commitments for sustainable use of those resources. Particularly those that are tenants, or simple occupants, of these ranches will have an adequate incentive to care for the resources if given "property" (though not dominial) rights over the natural resources.

2. Piura goat herders

2.1. Introduction

Piura is a semi-arid scrubland area on the northern Peruvian coast, where goats play a major role in the economy of peasants. Being a dry region, resources are scarce and of a strong unpredictable nature, which has led to marketing strategies that reflect a complex interplay of ecological, economic, and cultural factors.

2.2. Traditional forms of transactions

Perevolotsky (1986) studied the marketing strategies employed by goat herders in Piura, in conjunction with ecological and economic factors, using data from regional slaughterhouses. This author argued that decisions concerning livestock offtakes are the results of an "insurance" strategy emphasizing a build-up of herds during abundant years in order to compensate for considerable losses caused by frequent droughts. Perevolotsky (1986) argued that these goat keepers practice a "capital-manipulation strategy" in which goats and kids are used as a form of capital. Apparently this strategy is an optimal answer in a region characterized by sharp weather changes, with variations in precipitation and forage availability that strongly affect production, structure, and dynamics of the herds (Perevolotsky, 1986).

Perevolotsky (1983) analyzed the forms of transactions in Piura, and found that livestock transactions are usually institutionalized through the *al partir* arrangement (sharing system), in which the herder is responsible for the rancher's livestock and in return receives half of the kids produced by the herd during that period. The herder can also use the extra milk produced by the lactating goats after the kids

receive their share. The pastoralist is responsible for the amount of livestock he has received and has to prove, by providing the intact skins of the dead animals, that any loss of the rancher's animals was due to natural causes; otherwise he has to compensate the rancher with his own productive goats.

Perevolotsky (1983) found that the *al partir* arrangements resemble the reciprocal relations, locally known as *confianza* (trust), practiced by Mexican peasants (Dinerman, 1978, cited in Perevolotsky, 1983). Dinerman described reciprocal economic aid (mostly credit or cash loans) exchanged between non-kinsman males, heads of households, either members of the same community or of two different communities. Dinerman argued that such interpersonal relations are an adaptive response to a regional economy characterized by erratic supply, unpredictable demand, and price indeterminacy. Considering the nature of weather fluctuations and the unreliability of pasture in Piura, the analogy between the Mexican and the Piura peasants is clear, despite their involvement in different subsistence activities.

Curiously, Perevolotsky (1983) found that the *al partir* arrangement is also practiced in the opposite direction; herders leave small portions of their herd with ranchers residing in the cultivated zone. These transactions are obviously performed in times of forage shortage, when the herder is still not ready to, or is unable to, migrate from the desert. Usually the herder distributes only a few animals to each agriculturalist family because of the labor problems most of the ranchers face. Perevolotsky (1983) claimed that by dividing his herd among a couple of families in different agricultural sites, the pastoralist ensures long-term survival of at least a reproductive core of his herd.

2.3. New communal arrangements

These traditional and institutionalized arrangements of livestock transactions have been complemented with recent new communal organizations as a consequence of the land reforms that took place in Peru in 1969. While agrarian reforms, in general, affected large tracts of actual or potentially cultivated land, most pastures remained under state control. This is why in many parts of Latin America important numbers of herders still live on land owned and managed by the state. Perevolotsky (1985), while studying in depth the goat production systems of northern Peru, analyzed two communal systems of land tenure in the Piura valleys, different from the cooperatives: the "Grupo Campesino" (peasant group) and the "Comunidad Campesina" (peasant community). They show two different ways in which both historical social forces and ecological and economic factors have shaped present forms of land tenure and organization. It is worth describing Perevolotsky's (1985) results in some detail.

2.3.1. Grupo Campesino

The "Grupo Campesino" or "peasant group" was established by the Peruvian agrarian reform in those cases of agricultural estates divided into small fields, and given to the former workers. The socioeconomic organization of the group is supposed

to grant "campesinos" (peasants) some of the advantages enjoyed by communal units of production like cooperatives, e.g., credit, marketing, and technological advice. "Grupos campesinos" are found mainly in the highlands, where either ecological or social factors prevented the establishment of a cooperative. Horton (1973) called these groups pre-cooperative units.

However, the "grupos campesinos" of Piura are quite different. The agrarian reform office encouraged certain "ganaderos" (herdsmen), known locally as "arrendatarios" because they rented range from the hacienda, to organize themselves in order to get control over the communal resources of the non-agricultural former hacienda. The main problem in establishing this new social structure was that these people historically never formed an integrated social unit. This lack of a corporate unit was a result of the "divide and conquer" strategy employed by the patron in his relations with peasants; the nature of range subsistence, which called for small, isolated households; the absence of a communal social center; and the demographic mobility of the population. Given these factors, the loose social structure of goat herders is easily understood.

With the help of government development agencies, herders from the former lumber haciendas were organized into groups with a structure similar to that of agricultural cooperatives: management committees elected by all the group's members. The main difference between an agricultural cooperative and a peasant group is that the former officially owns the means of production while the latter does not. Most "grupos campesinos" were promised that, if they succeeded in managing their communal affairs well, their status would be elevated to that of livestock cooperatives.

In the former lumber hacienda Jahuay Negro, near the Ecuadorian border in the province of Sullana, a group of "ganaderos" had succeeded in obtaining official control over the rangeland they used (48,145 ha). However, due to a slow and complicated legal process the "grupo campesino" (the official owner and user of the land) became involved in land disputes, particularly with the governmental agency in charge of civil development (ORDENORTE); most of the disputes were related to damage to their economy and the violation of their property as a result of ORDENORTE leasing, e.g., lumber rights to private entrepreneurs. In this controversy ORDENORTE justified its action by claiming that the "grupo campesino" owned only the land but not the lumber on it. Fortunately for the "grupo campesino," it was discovered that most of the money from this initiative ended up in the pockets of ORDENORTE officials, so the whole operation was halted (El Tiempo, 1982, cited in Perevolotsky, 1985).

Perevolotsky (1985) believed that decision-making processes in "grupos campesinos" are marked by three conspicuous considerations: (1) the demographic mobility of group members, so that communication and efficient contact become practical obstacles to the organization of an active socio-political group; (2) the inexperience of local leaders in handling public affairs (contacts with outsiders, passing decisions in the general assembly, etc.); and (3) the unwillingness of individual

"ganaderos" to give up their own interests on behalf of the group interest (probably a remnant of intra-group competition and conflict during "hacienda" times).

Despite problems in organizing the political process, some peasant groups have made encouraging decisions regarding resource management. For example, Perevolotsky (1985) referred to a communal agreement that no member would cut timber or firewood in commercial quantity, and that even cutting wood to build or repair one's house, as well as clearing a "despoblado" (scrublands) plot for agriculture, first had to be approved by the administrative committee. New members and even relatives of veteran members were accepted into the group only upon presenting a letter of recommendation from authorities in their previous residential area. Finally the group could, and sometimes did, expel members because of abuse of communal or private property.

The "grupo campesino's" management of natural resources seems more efficient and rational than the lumber "haciendas," mainly because all members of the group share a single economic interest: the welfare of their livestock. Collectively, the "grupo campesino" now forbids any commercial lumbering. It also forbids non-members' cattle or goat herds in its territory, thus minimizing the threat of overexploitation. Since little intragroup cooperation is required among members, and since any individual concessions are compensated by a contribution to goat herding, the individual's main source of subsistence, this political structure seems less liable to the individual/group conflicts so characteristic of cooperatives (Perevolotsky, 1985).

Still the principal problem of the "grupos campesinos" is insecure land tenure. The fact that they do not officially own the land makes them doubt whether they will be able to maintain long-term control over the resources and to prevent reencroachment by new patrons. From these herders' point of view, only official title to the land and its resources would provide a permanent solution to such potential problems. Many members still question whether a communal organization like a cooperative or a loose, individualistic political structure might not be better.

2.3.2. Comunidad Campesina

The "Comunidad Campesina" (peasant community), previously known as "comunidad indígena," is a remnant of the traditional Indian community structure. The communities control the basic means of production: water, land, and pasture. They usually allocate individual cultivation rights to communal lands and attempt to regulate water or pasture utilization in a way most beneficial to all members. In two of the more "pastoral" communities of Olmos and Salas, Perevolotsky (1985) analyzed the situation of herdsmen. The problem there was a very old one: the communities had difficulty in maintaining exclusive control of their vast natural pastures since a dispute that arose in the 17th century between the Indians and ranchers who grazed their livestock on community rangelands without permission. The livestock not only depleted the community's forage resources but also caused damage to agricultural fields and irrigation canals.

Finally in 1893 a decision was made to allocate communal land to individuals ("comuneros") that received a parcel of four "cuadras" (23,000 m^2), fenced it in, and made it a "potrero" (an enclosure in which pasture could be kept for the dry season). If water was available, portions of the parcel were also dedicated to temporary agriculture. The community ruled that when the fence collapsed of age, the land would return to community control. "Comuneros" could not sell their "potreros." The rest of the community's territory, usually rangeland of secondary quality, remained communal property with free access for all community members. This system of land tenure has continued to present times (Perevolotsky, 1985).

Literature Cited

Ares, J., A. M. Beeskow, M. Bertiller, M. Rostagno, M. Irisarri, J. Anchorena, G. DeFossé & C. Merino. 1990. Structural and dynamic characteristics of overgrazed lands of northern Patagonia, Argentina. Pp. 149–175 *in* A. Breymeyer (editor), Managed Grasslands. Elsevier Science, Amsterdam.

Barbería, E. M. 1996. Los Dueños de la Tierra en la Patagonia Austral, 1880–1920. Editorial Universidad Nacional de la Patagonia Austral. Río Gallegos, Prov. de Santa Cruz, Argentina.

Grigera, D. E. & E. H. Rapoport. 1983. Status and distribution of the European hare in South America. J. Mammal. 64: 163–166.

Horton, D. 1973. Haciendas and Cooperatives. Research Paper No. 53. Land Tenure Center, University of Wisconsin, Madison.

Howard, W. E. & J. N. Amaya. 1975. European rabbit invades western Argentina. J. Wildlife Management 39: 757–761.

Jaksic, F. M. & J. L. Yáñez. 1983. Rabbit and fox introductions in Tierra del Fuego: History and assessments of the attempts at biological control of the rabbit infestation. Biol. Conserv. 26: 367–374.

López, H. 1998. El zorro colorado, una señal de alarma. Interview in the newspaper "La Nación," July 11, 1998. Section 5, page 6.

Perevolotsky, A. 1983. Herder-Rancher Relationships in the Tropical Desert of Piura: The Role of Uncertainty and Variable Environment. Paper presented at the 11th Congress of the Anthropological and Ethnological Sciences, Vancouver, August 20–25, 1983.

Perevolotsky, A. 1985. Inhabitants of the uninhabited region: Goat herders in Piura, Peru. Technical Report Series Number 33, Small Rumiant Collaborative Research Support Program. 174 pp. University of Missouri-Columbia.

Perevolotsky, A. 1986. To sell or not to sell—A pastoralist's dilemma: A lesson from the slaughterhouse. Human Ecology 14(3): 287–310.

Redford, K. H. & J. F. Eisenberg. 1992. Mammals of the Neotropics. Vol. 2. The Southern Cone: Chile, Argentina, Uruguay and Paraguay. Univ. Chicago Press, Chicago.

Soriano, A. & J. M. Paruelo. 1990. El pastoreo ovino: Principios ecológicos para el manejo de los campos. Ciencia Hoy 2: 44–53.

Missouri Botanical Garden Library
3 1753 00284 9120

ISBN 0-915279-85-1